Integrating conservation with agricultural production: linear strips of native vegetation support declining woodland birds and provide benefits to pasture

A thesis submitted for the degree of
Bachelor of Advanced Science (Hons)

Brodie Crouch

October 2022

School of Earth and Environmental Sciences
The University of Queensland, St Lucia Campus, Queensland

Preface for the examiners

This dissertation has been written in a format suitable for the submission to the journal *Agriculture, Ecosystems and Environments* (AEE). Because of this, I have included this brief preamble to clarify for the examiner:

1. the broader context of the research including the research programme in which it is aligned,
2. the development background and overall research development,
3. the word limit constraint and how I have addressed it, and
4. to acknowledge the various roles of academic supervisors and others and clarify where and how they have supported the research.

In recent years, there has been an increasing trend for Honours students to submit their dissertation in the format of a peer reviewed journal. This is particularly important for those students who seek enrolment in a higher degree research programme. However, AEE has a word limit of 8, 000 words, which is far lower than the normal expectation for a 12 # thesis for the School of Earth and Environmental Science, which would normally range between 12,000 and 25,000 words. This word limit constrains the extent to which a student can engage more broadly with literature that supports the context of this work. To this end, I have included an extended introduction and discussion which brings the thesis above the 8,000 word limit, but still below the expected 12,000 – 25,000 words.

This dissertation describes the results of field work conducted on grazing properties over four separate trips throughout 2022. Southwest Queensland properties were identified by myself via satellite imagery and prioritized by the abundance of linear strips of brigalow dominated vegetation and proximity to one another. Final property selection was conducted on-ground, prior to field work commencing after contacting landholders to discuss their involvement.

Previous research had identified that whilst the biodiversity outcomes of many agricultural land conservation strategies were well studied, few also quantified the impact of the strategies on agricultural production (Bianchi et al. 2013). Lack of information around the impacts to agricultural productivity restrict uptake, even if the conservation benefits are clear (Pannell et al. 2006). Hence, this thesis aimed to investigate the effect of brigalow-dominated linear strips on woodland bird communities and pasture productivity.

Numerous studies had documented woodland birds using linear strips in other Australian landscapes (Bentley & Catterall 1997; Bowen et al. 2009; Hall et al. 2018; Saunders & Hobbs 1991), however, none covered my study area; despite linear strips being relatively common in the landscape at a range of widths, orientations, landscape scale factors etc. This allowed analysis of how these differences may drive changes in the woodland bird communities that utilize them.

Although several studies across Australia have documented the effects of strips on crop and pasture productivity, results vary substantially between study regions and contexts (Bird 1998; Bird et al. 2002; Cleugh et al. 2002; McKeon et al. 2008; Meinke et al. 2002). This variation meant that context-specific information was critical, and extrapolating research from other Australian agricultural systems to my study area would not be appropriate. To the best of my knowledge, only one study had investigated the effects of brigalow-dominated strips on pasture productivity, and it recommended that “further study on tree strips of different orientation, location and year-types is warranted to build up a larger database” (McKeon et al. 2008, pg. 4). Prior to my study, no other research appeared to have followed through on this avenue for further work, so I aimed to further this knowledge specific to my study area.

Contributors from different affiliations helped to make this Honours thesis possible. I have summarized the role of each contributor in the table below.

Name and affiliation	Contributions
Brodie Crouch (UQ)	Literature review, experimental design, site selection, establishing and maintaining landowner contacts, field work logistics, bird, pasture and vegetation surveys, statistical analysis of bird and vegetation data, writing, editing.
Prof. Martine Maron (UQ)	Advice on experimental design, assistance with bird and vegetation surveys, editing, supervision.
Dr. Hayley McMillan (Department of Agriculture and Fisheries (DAF))	Advice on experimental design, editing, supervision, fieldwork, establishing landowner contacts.
Dr. Bradd Witt (UQ)	Advice on experimental design, editing, supervision.
Joshua Peart (DAF)	Field work logistics, bird, pasture and vegetation surveys.
David Mayer (DAF)	Support for statistical analysis of pasture data, editing.
Dr Steven Bray (DAF)	Advice on experimental design.
Annie Kelly (Queensland Herbarium)	Provision of data from Biocondition model.
Dr. Jason Barnetson (Department of Environment and Science)	Advice on experimental design.

Integrating conservation with agricultural production: linear strips of native vegetation support declining woodland birds and provide benefits to pasture

B. Crouch, H. McMillan, J. Peart, B. Witt, D. Mayer, A. Kellie, M. Maron

ABSTRACT

Linear strips of native vegetation are common features of agricultural landscapes. They may be retained on farms for the benefits they can provide to the production system. These strips can also provide important habitat resources for a variety of taxa. Substantial scope remains to increase the number and extent of these strips, yet their uptake as an on-farm conservation measure is restricted by uncertainty surrounding both their impacts on production, and also their habitat value to taxa of conservation concern. Here we ask whether a declining woodland bird community uses strips of native vegetation retained in otherwise-cleared grazing landscapes, and explore the characteristics of strips that influence woodland bird species richness and abundance within the strips. We quantify the impact that strips have on adjacent pasture production in the highly-modified brigalow *Acacia harpophylla* landscapes of southern Queensland, Australia. We surveyed the bird communities of 47 sites within retained strips ranging from 30 – 388 m in width, and sampled pasture basal area along transects perpendicular to the strip edge. We found that brigalow-dominated strips were used by declining woodland birds. Using generalised linear mixed models, we found that the abundance of woodland birds was greater at sites in wider strips, although species richness was not affected by strip width. However, the species richness of woodland birds was negatively correlated with abundance of native but aggressive honeyeaters of the genus *Manorina*. Five of eight pasture survey transects showed strong evidence of a positive effect of strips on pasture basal area. These results suggest that the retention of both remnant and regrowth brigalow-dominated vegetation in the form of strips could be an effective vegetation management strategy that delivers improved outcomes for woodland birds whilst minimising pasture productivity losses.

Key words: woodland birds, brigalow, linear habitats, grazing lands, pasture productivity.

INTRODUCTION

Conservation on privately-owned agricultural land is necessary to preserve and improve biodiversity beyond the boundaries of protected areas (Kearney et al. 2022). However, the widespread uptake and adoption of conservation measures on private land is strongly influenced by the need to improve or maintain the economic resilience of agricultural businesses (Bianchi et al. 2013). Furthermore, conservation measures need to align with the personal goals and values of landowners, or they struggle to gain widespread traction and uptake (Pannell et al. 2006). Hence, it is important to understand not only the conservation benefits of such measures, but also their impacts on agricultural production (Scherr & McNeely 2008).

The retention of native vegetation in the form of linear strips, such as shelterbelts, windbreaks, hedgerows, roadside verges and corridors, is common throughout the world's agricultural landscapes (Brandle et al. 2004; Mayrinck et al. 2019; Saunders & Hobbs 1991; Sullivan et al. 2017). These linear strips can benefit native flora and fauna whilst also providing production benefits (Ryan et al. 2010). For example, such strips can provide shelter and protection for livestock and crops (Bird et al. 1992). Shade and shelter can improve livestock productivity by reducing heat stress (Edwards-Callaway et al. 2021), and microclimates created in the lee of strips, and protection from temperature and wind extremes, can improve crop and pasture yields (Cleugh et al. 2002; McKeon et al. 2008).

Despite their benefits, strips can impose several costs on agricultural businesses, and these costs can limit their popularity and uptake (Rempel et al. 2017). First, competition between trees and adjacent crops or pasture for light, nutrients and water can result in productivity losses (Bennell & Verbyla 2008; Scanlan 1991). The net impact of strips on yield varies substantially among studies, production systems and environmental contexts (Cleugh et al. 2002). Second, strips may provide habitat for pest fauna, which may cause crop damage and compete with livestock for pasture (Jamieson et al. 2002; McAlpine et al. 1999). For example, in Australia, strips can provide shelter for macropods that are hyper-abundant, and that add to total grazing pressure (McAlpine et al. 1999). Finally, strips may reduce property management efficiency; for example, a decline in hedgerows and shelterbelts in Europe and Canada has been driven by the greater efficiency of managing larger, uninterrupted fields (Mayrinck et al. 2019; Sklenicka et al. 2009).

If production losses associated with linear strips are not compensated for by beneficial effects on production, then, in the absence of other incentives, retention or regeneration of strips will be at best marginal (Pannell et al. 2006). However, the emergence of environmental markets and biodiversity certifications highlights a shift towards valuing agricultural land not only for agricultural production, but also for other environmental values (Swinton et al. 2007). Since strips also serve to provide important habitat resources for a variety of taxa, schemes that provide payments for ecosystem services and access to premium markets may present incentives for land managers to retain and, where possible, increase habitat in the form of linear strips.

Linear strips in agricultural landscapes increase habitat heterogeneity and provide important resources for taxa including birds, insects, reptiles and mammals (Barth et al. 2020; Guiller et al. 2022; Smith et al. 2015; van der Ree & Bennett 2003). For example, hedgerows are often used by birds for nesting (Hinsley & Bellamy 2000), and strips of grassland kept in cultivated fields can provide important winter food resources for a variety of taxa (Šálek et al. 2022). In agricultural regions of northern Victoria, linear strips along riparian zones and roadside verges also provide important habitat for woodland-dependent birds (Hall et al. 2018). Furthermore, strips can provide habitat for natural enemies of crop pests, and sequester carbon on-farm (Gagic et al. 2018; Mayrinck et al. 2019). Strips in agricultural systems vary in several ways, including their width, composition (such as being remnant or regrowth vegetation), vegetation density, height, and ground layer intactness, all of which can affect their use by different taxa (Graham et al. 2018; Green et al. 1994; Hinsley & Bellamy 2000; Jellinek et al. 2014; Usieta et al. 2013). For example, wider strips may support both greater species richness (hereafter richness) and abundance of woodland birds per unit area than thinner strips (Bonifacio et al. 2011; Hall et al. 2018). Thinner strips may be more prone to negative edge effects, with nest predation rates shown to be higher along edges than interior habitat (Piper & Catterall 2004), and some species showing substantial edge avoidance (Zurita et al. 2012). Despite this, thin strips have been shown to support a similar richness of woodland-dependent birds compared to larger, more intact patches (Bowen 2009), and would likely be a more attractive option to landholders since they likely represent a lower cost option in productive agriculture systems (McKeon et al. 2008).

Here, we focus on voluntarily retained strips and roadside verges in grazing landscapes of a nationally-listed endangered ecological community in Queensland, Australia – brigalow *Acacia harpophylla* forest and woodland *(Environment Protection and Biodiversity Conservation Act 1999)*. Originally common and widespread throughout eastern Australia's large Brigalow Belt bioregion, brigalow ecosystems have declined to less than 10% of their original extent due to a large-scale, government initiated agricultural development scheme leading to extensive land clearing in the second half of the 20th century (Accad et al. 2022; Seabrook et al. 2006). The loss of brigalow and other ecosystems from the Brigalow Belt bioregion (hereafter, BBR) has driven local extirpations and substantial faunal population declines for many species (Ponce Reyes et al. 2016). For example, woodland-dependent birds have suffered substantial declines in the BBR, having being negatively affected by habitat loss and fragmentation, and biotic interactions with hyperaggressive competitors (Bowen et al. 2009; Collard et al. 2008; Maron, Goulding, et al. 2012; Maron et al. 2013; Woinarski et al. 2006).

Despite the extensive BBR habitat losses, there are also important opportunities for its restoration. Once cleared, brigalow generally regrows from suckers, and is generally repeatedly re-cleared to maintain open pasture (Scanlan & Anderson 1981). As such, at any given time, considerable areas of secondary regrowth vegetation of different ages occur throughout the BBR (Lucas et al. 2014). This regrowth could be strategically retained to increase the number, extent and width of strips. Secondary forests in many parts of the world hold similar potential to restore both biodiversity and terrestrial carbon stores in agricultural landscapes (Bowen et al. 2007; Chazdon et al. 2009; Dwyer et al. 2009; Heinrich et al. 2021; Mukul et al. 2016). These forests could offer an opportunity to restore numerous ecosystem services whilst minimizing production losses, and contribute to industry objectives such as the Australian red meat industry's 'Carbon Neutral by 2030' target (Meat and Livestock Australia 2020; Ryan et al. 2010; Young 2017).

It remains unclear whether retaining or increasing linear brigalow-dominated strips offers the opportunity to support woodland bird populations in the BBR without compromising agricultural production. Only two studies have examined the effect of brigalow strips on adjacent crop and pasture production, with mixed results; one found no evidence of beneficial effects (Bradley 2007), whilst the other reported substantial beneficial effects on pasture adjacent to the strips (McKeon et al. 2008). Furthermore, several studies have examined how small and sometimes linear patches of brigalow influence woodland-dependent bird communities in the eastern part of the BBR (Bowen 2009, Collard et al. 2008; Maron et al. 2012). These have found that even small patches of brigalow in agricultural matrices provide a substantial contribution to the avifaunal diversity of the landscape (Collard et al. 2008), that narrow linear remnants support similar woodland-dependent bird richness to brigalow patches within larger conservation reserves (Bowen 2009), and that an area-sensitive woodland-dependent bird exhibits physiological signs of long-term stress in landscapes with less remnant woodland (Maron et al. 2012). However, no studies to date have examined the impact of brigalow strips on woodland-dependent birds and pasture in tandem, nor the effect of strip width on woodland birds. Furthermore, none has examined the western part of the BBR, where lower rainfall and less-productive soils may affect both the conservation value of strips and their impact on production.

In this paper, we address this gap in knowledge by examining the extent to which woodland-dependent birds use brigalow-dominated strips of different widths in the western BBR. We examine what aspects of the strip and the surrounding landscape affect this use, and the effect of strips on adjacent pasture. We address the following questions:

1. Are strips of brigalow-dominated vegetation in fragmented grazing landscapes used by woodland-dependent birds?
2. What characteristics of these strips influence the richness and abundance of woodland-dependent birds?
3. How do strips affect pasture basal area (as a proxy for pasture yield) in adjacent pasture?

METHODS

Study Area

The study region encompassed an area of approximately 1300 km^2, covering the Morven, Augathella and Mungallala regions (Figure 1). The region is close to the border of the semi-arid and subtropical climate zones, and experiences average daily temperature ranges of 20–35°C in summer, and 4–22°C in winter (Bureau of Meteorology 2022a). Average annual rainfall at Augathella is ~500 mm (Bureau of Meteorology 2022b). The study area occurs within the Southern Downs sub-region of the BBR, at its western extent. Dominant vegetation communities of the study region include brigalow (*Acacia harpophylla*) dominated open forests/woodlands on clay soils, poplar box (*Eucalyptus populnea*) communities on alluvial plains and hill footslopes, and white cypress pine (*Callitris glaucophylla*) woodlands, on Cainozoic alluvial plains (Neldner et al. 2019).

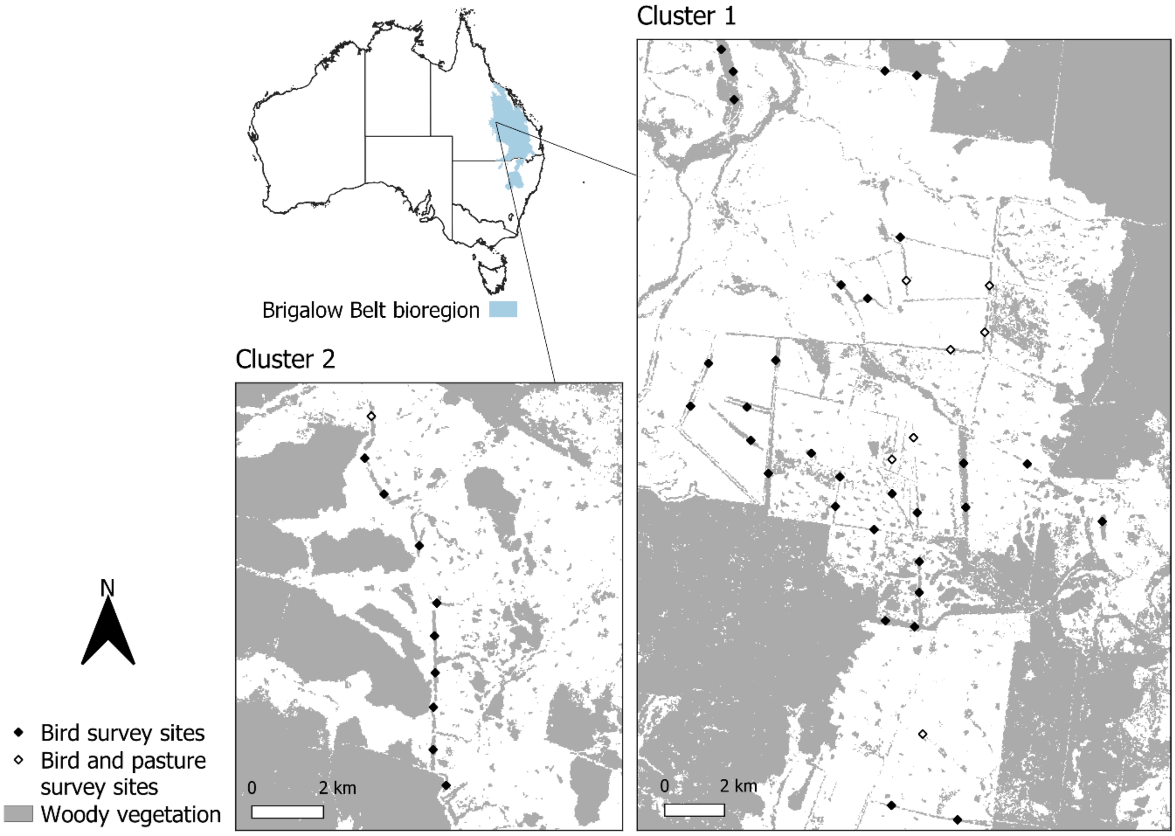


Figure 1 Location of the survey sites within the Brigalow Belt bioregion, Australia. Sites were located in two clusters approximately 30 km apart. Woody vegetation cover was sourced from Department of Environment and Science, 2015.

Since the development of the region for agriculture, the extent and connectivity of remnant vegetation has substantially decreased, with only 31.4% of the Southern Downs sub-region consisting of remnant vegetation as of 2019 (Department of Environment and Science 2022c). Brigalow-dominated vegetation has been preferentially cleared due to the higher soil fertility of land that brigalow typically grows on (Seabrook et al. 2006). As a result, brigalow-dominated vegetation on heavy clay soils has been reduced to 12.2% of its former extent (Neldner et al. 2019). This remainder is highly fragmented and often occurs as isolated linear strips in an agricultural matrix (Dwyer et al. 2009). This landscape change is relatively recent, having occurred within the second half of the 20th century (Seabrook et al. 2006). Land use primarily consists of grazing of improved and native pastures (Ponce Reyes et al. 2016).

Effect of strips on woodland bird communities

Site selection

Bird and vegetation surveys were conducted at 47 study sites located in brigalow-dominated strips across four cattle grazing properties. To identify potential sites, we initially used Google Earth imagery combined with Queensland Globe Pre-clearing BVG 1:1 million mapping (Queensland Globe 2022). Potential survey sites, located at least 800 m apart, were selected to comply with the following criteria:

- *Sites classified as Broad Vegetation Group 25a* (Open forests to woodlands dominated by *Acacia harpophylla* (brigalow) sometimes with *Casuarina cristata* (belah) on heavy clay soils) (Neldner et al. 2019).
- *Sites in vegetation that met the definition of a 'strip'.* A strip was defined as an area of woody vegetation whose length was at least three times its width, with both of its long edges separated from woody vegetation for at least 80% of the strip's total length. Strips were deemed separate if there was a gap of at least 50 m between them.
- *Sites located at least 500 m from farm dams, other bodies of permanent water, and houses.*
- *Sites located in strips ≥30 m wide* to allow for the dimensions of the bird and habitat survey plots.

Sites were selected to represent a gradient of strip widths (Figure 2). Within suitable strips, sites were haphazardly located, although care was taken to ensure that strip width was not obviously confounded with within-strip canopy cover and habitat cover within a 1 km radius of each site. Sites were chosen to minimize variation in elevation, soil type, rainfall, and vegetation composition. A site scoping trip was conducted in June 2022 to confirm ease of access, vegetation type, and landholder permission. Each site comprised a 1 ha bird survey plot. The dimensions of the plot varied among sites according to width. For strips between 30–50 m in width, bird survey plots were 30 x 330 m. For strips >50 m in width, bird survey plots were 50 x 200 m.

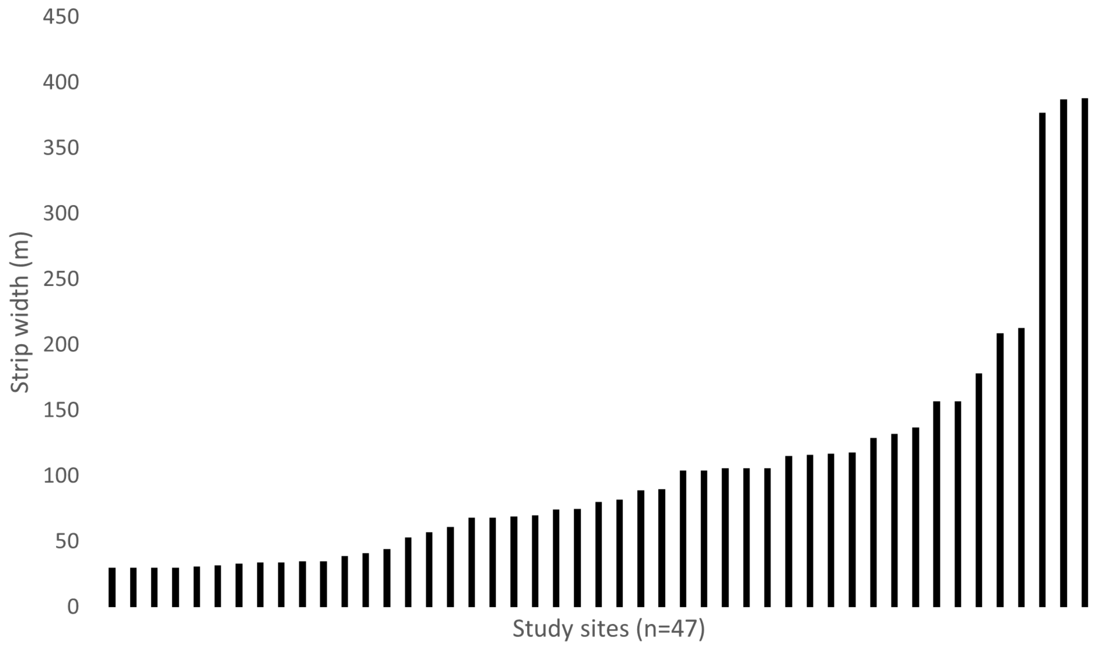


Figure 2 The width of strips within which the 47 sites were located.

Response variables

Each site was surveyed for birds twice during 2022, once during winter (July/August), and once during spring (September). Surveys were done within 5 hours after sunrise, and were not done during periods of strong wind, rain, or unusually hot (>32 degrees Celsius) or cold (<0 °C) temperatures. BC, JP and MM conducted the bird surveys. During each survey, the surveyor walked the centreline of the 1 ha survey plot and recorded all birds present within the plot and below the canopy. Birds detected within the strip but outside of the survey plot were noted but not included in the analysis.

We used the classification of Bowen et al. (2009) to classify all species detected as either ‘woodland-dependent’ or ‘other’. This classification defined species as being woodland-dependent ‘*if they primarily forage, breed and shelter within woodlands and are rarely observed in crops or pasture*’ (Bowen et al. 2009, p. 3052). This allowed us to define the following response variables: woodland-dependent bird (hereafter ‘woodland bird) species richness and abundance. Abundance was recorded as the sum of detections across the two repeat surveys, whilst richness was the total number of unique species recorded over the two surveys. In addition, the total abundance of *Manorina* honeyeaters was calculated as a response variable by summing the number of yellow-throated (*M. flavigula*) and noisy miners (*M. melanocephala*) detected per plot.

This research was conducted with the approval of The University of Queensland’s Native/Exotic Wildlife and Marine Animals Animal Ethics Committee 2022/AE000361.

Explanatory variables

Local and landscape-scale variables that may influence woodland bird communities were measured at each site using several survey techniques (Figure 3 and Table 1). Local-scale variables, including those describing vegetation structure and floristics and other habitat elements (full details in Table 1), were measured within two 50 x 10 m vegetation survey plots that were located within the 1 ha bird survey plot (Figure 3). The long edge of the subplots were parallel to the edge of the strip, and habitat variables recorded within the two subplots were averaged to give one measure of each habitat variable per site. Within these 50 x 10 plots, a 50 m line transect, two 20 x 5 m subplots and ten gridded 0.5 x 0.5 m quadrats were established, within which different variables were measured (Table 1). The extent of habitat cover surrounding each site was measured within a 1 km radius, using remnant vegetation and high value regrowth mapping (Department of Environment and Science 2022a, 2022b). These two vegetation classes were defined using the statutory definitions provided by the Queensland’s government (Department of Natural Resources, Mines and Energy 2019).

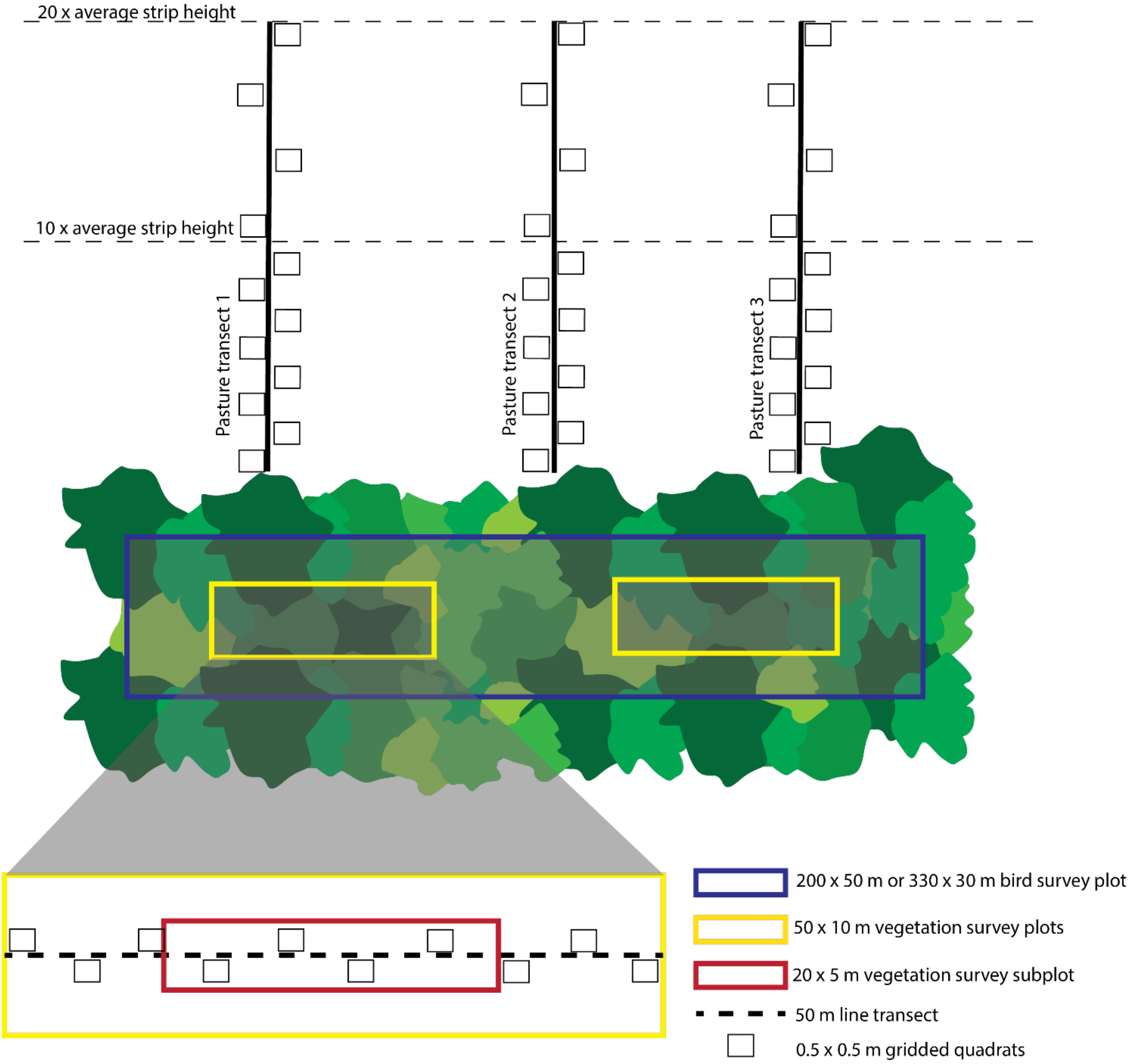


Figure 3 A stylised representation of the survey site, showing the bird survey plot, the subplots within which various vegetation and habitat features were measured, and the adjacent pasture transects. Note the number of quadrats shown along the pasture transects do not reflect the actual number of quadrats measured in the field, and instead show the differing sampling intensity between 0 - 10 H and 10 - 20 H.

Table 1 Summary of explanatory variables measured at each site

Variable	**Description**	**Survey area**
Local-scale		
Width (m)	Three measures of strip width taken along the length of the bird survey plot, then averaged.	Bird survey plot
Vegetation status	A categorical variable, defining whether the site was located in a remnant or regrowth (post-clearing) strip. This was based upon land clearing history as communicated by land-owners.	Bird survey plot
Manorina spp. abundance (count)	Total number of Noisy and Yellow-throated Miners recorded within each bird survey plot over the two repeat surveys.	Bird survey plot
Ecologically distinct canopy (EDC) cover (%)	The percentage cover of the layer making the greatest contribution to above-ground biomass. A surveyor (BC) walked along the 50 m transect and recorded the vertical projection of the EDC over the transect line.	50 m line transect
Emergent cover (%)	The percentage cover of the tallest layer above the EDC. The same method was used as for the EDC.	50 m line transect
Subcanopy cover (%)	The percentage cover of vegetation that were not classified as shrubs, but that were below the EDC. The same method was used as for the EDC.	50 m line transect
Shrub cover (%)	Percentage cover of vegetation < 2 m tall, or multi-stemmed from base. The same method was used as for the EDC.	50 m line transect
Vegetation structural complexity	Shannon's diversity index of emergent, EDC, subcanopy and shrub cover.	50 m line transect
Small shrubs (count)	Stem count of shrubs < 2 m tall, distinguishing between brigalow and non-brigalow.	20 x 5 m subplot
Large shrub count (count)	Stem count of shrubs > 2 m tall, distinguishing between brigalow and non-brigalow.	20 x 5 m subplot
Sapling count (count)	Stem count of saplings (vegetation > 2 m tall with DBH < 2.5 cm)	20 x 5 m subplot
Small tree count (count)	Stem count of small trees (vegetation > 2 m tall with DBH >2.5 cm and < 15 cm).	20 x 5 m subplot
Medium trees (count)	Stem count of medium trees (vegetation > 2 m tall with DBH >15 cm and < 25 cm).	20 x 5 m subplot
Vegetation density (count)	Sum of small and large shrub, sapling, small tree and medium tree count, calculated from the five variables above.	20 x 5 m subplot
Large trees (count)	Stem count of large trees (vegetation > 2 m tall and DBH > 25 cm)	Bird survey plot
Dead trees (count)	Stem count of dead trees (dead vegetation > 2 m tall and DBH > 25 cm).	Bird survey plot
Coarse woody debris (count)	Count of fallen timber longer than 30 cm and diameter > 2.5 cm.	20 x 5 m subplot
EDC tree height (m)	Average height of the EDC measured with laser rangefinder.	50 x 10 plot
Basal area (m^2/ha)	Basal area of woody vegetation using 'Bitterliech' sweep technique (Grosenbaugh 1952).	Gridded quadrats
Leaf litter cover (%)	Average percentage of leaf litter cover within 20 0.25m^2 quadrats	Gridded quadrats
Bare ground (%)	Average percentage of bare ground within 20 0.25m^2 quadrats	Gridded quadrats
Buffel grass cover (%)	Average percentage of buffel grass cover within 20 0.25m^2 quadrats (10 quadrats per 50x10 m subplot).	Gridded quadrats
Sward height (cm)	Average height of grass sward within 20 0.25m^2 quadrats.	Gridded quadrats
Mistletoe abundance (count)	Count of mistletoe, noting flowering or fruiting.	50 x 10 plot
Gilgai presence (rank)	Score 0 – 3: (0=absent; 1= light, 2 = medium; 3 = heavy)	50 x 10 plot
Landscape scale		
Remnant and high value regrowth cover (%)	The percentage cover of remnant or high value regrowth in a 1 km radius around each bird survey plot.	1 km radius around survey site.

From this list of potential variables, we selected five key explanatory variables (strip width, buffel grass cover, abundance of *Manorina* spp., vegetation growth status and surrounding habitat cover) to use in our modelling of woodland bird richness and abundance. First, strip width has been an important explanatory variable for woodland birds in other studies focusing on linear strips in agricultural landscapes (Bonifacio et al. 2011; Hall et al. 2018). Second, buffel grass cover was chosen since it can affect the fire dynamics of brigalow woodland, which can lead to degraded stands of brigalow (Butler & Fairfax 2003), and can also influence the foraging behaviour of native birds (Young, L & Schlesinger 2015). For example, elsewhere in Australia, ground-foraging woodland birds have been shown to avoid sites where the ground layer has been invaded by exotic pasture grasses (Maron & Lill 2005). Third, the abundance of *Manorina* spp. was selected since these birds aggressively defend territories, and their abundance has been associated with a reduction in both the richness and abundance of small birds in multiple studies (Eyre et al. 2009; Hall et al. 2018; Maron et al. 2013). Fourth, regrowth woodlands often provide different habitat resources to remnant woodlands, and may be used by a different suite of species (Bowen et al. 2009). Additionally, any increase in the extent of linear strips in the BBR would necessarily be facilitated by regrowth retention, so we included vegetation status (remnant or regrowth) in our models. Finally, numerous studies have found a positive effect of the amount of surrounding habitat cover of either remnant or regrowth vegetation within the landscape surrounding sites on woodland bird richness and abundance (Bowen et al. 2009; Haslem et al. 2021). As such, we used cover of remnant and high value regrowth within 1 km of sites as an explanatory variable.

For the models of *Manorina* spp. abundance, we chose five explanatory variables. First, we used pasture sward height as an indicator of grazing intensity, since *Manorina* honeyeaters have been shown to be more abundant in sites with greater grazing intensity (Eyre et al. 2009) . Second, vegetation density was included since sites with more understory and midstory are less likely to be dominated by *Manorina* honeyeaters (Eyre et al. 2009; Hastings & Beattie 2006). Third, the amount of remnant and high value regrowth in the surrounding landscape was included since *Manorina* spp. tend to be more abundant in more fragmented landscapes with reduced habitat cover and more edges (Maron et al. 2013; Piper & Catterall 2003; Thomson et al. 2015). Fourth, the amount of coarse woody debris was incorporated in the model since Eyre et al. (2009) found that coarse woody debris had a significant negative effect on *Manorina* spp. abundance. Finally, we included strip width since *Manorina* spp. have been shown to preferentially use edge habitat and may therefore be more common in thinner strips, which have a higher edge-to-interior ratio (Clarke & Oldland 2007; Major, Christie & Gowing 2001).

Woodland bird and Manorina *spp. data analysis*

All woodland bird statistical analyses were conducted using R version 4.2.1 (RStudio Team 2022).

To examine how woodland bird richness and abundance, and *Manorina* spp. abundance varied with habitat measures, we used generalized linear mixed models (GLMMs) followed by model averaging to estimate the effect size and relative importance of each explanatory variable. We used the R package lme4 to fit the GLMMs (Bates et al. 2015). We assumed a Poisson distribution as our response variables were count data, and we tested the GLMMs for overdispersion. We included 'Property' as a random variable since our survey sites were clustered on four different properties. In addition, we also included 'Strip ID' as a second random variable since some survey sites were located within the same strips (although at least 800 m apart). Prior to modelling, we checked for collinearity among explanatory variables using Pearson (for interval scale variables) and Spearman rank (for ordinal scale variables) correlation coefficients. Only one from any pair of predictors with correlation coefficient of >0.6 were included in our models.

The model structures were:

1. Woodland bird richness or abundance = intercept + Buffel grass cover + *Manorina* spp. abundance + Strip width + Vegetation status + Remnant and high value regrowth cover + (1|property) + (1|strip ID)
2. *Manorina* abundance = intercept + Buffel grass cover + Strip width + Remnant and high value regrowth cover + Vegetation density + Coarse woody debris + (1|property) + (1|strip ID)

We originally intended to include vegetation structural complexity as one of the explanatory variables, since a higher diversity of strata, particularly of understory and midstory vegetation, can improve habitat suitability for some woodland birds (Lindenmayer et al. 2010), and may result in greater woodland bird abundance (Eyre et al. 2009). However, the correlation coefficient between vegetation structural complexity and vegetation status was >0.6, so it was excluded from the model.

The package MuMIn (Bartoń 2022) was used to compare the performance of all possible model subsets of the global model using Akaike's information criteria adjusted for small sample size (AICc). Model averaging reduces the sensitivity of coefficient estimates to which particular variables are included in a given multiple regression model (Burnham & Anderson 2002). We applied model averaging, based on AICc weights, across all models within 4 AICc values of the best performing model to generate a set of coefficients, standard errors, and 95% confidence intervals for each of those variables. Explanatory variables whose confidence interval did not intersect zero were treated as having an important influence on the woodland bird response variables. The goodness of fit of each global model was characterised using a marginal R^2. We estimated model uncertainty by examining the number of models within two AICc values of the best model, and whether the null model was present within four AICc values of the best model.

Finally, we explored how six key habitat attributes varied with strip width , in order to understand potential mechanisms through which strip width might influence bird assemblages. We fitted linear and non-linear models of the following forms:

1. $Response = (a \times strip\ width) + b$
2. $Response = a \times \ln width + b$
3. $Response = a \times width^{c}$

Where response was one of the six strip characteristics examined (mistletoe abundance, large trees, vegetation structural complexity, buffel grass cover, vegetation density and coarse woody debris) and *a* was the regression coefficient, *b* the intercept, and *c* the power term.

We then used AICc to compare model fit, and for the most parsimonious model of each variable, used two-sided T-tests to identify significant relationships. P-values < 0.05 were deemed significant.

Effect of strips on adjacent pasture

Site selection and survey design

Pasture surveys were conducted adjacent to a subset of eight strips selected for bird surveys. All pasture surveys were conducted during the July/August 2022 field trip, when pasture was senescent. At each strip, we surveyed pasture along a minimum of three pasture transects, which started at the edge of the strip and ran perpendicular to the strip (Figure 3). The distance from the strip edge was expressed in terms of multiples of tree height based on recommendations by multiple studies focusing on shelterbelt effects on adjacent crops and pasture (Cleugh et al. 2002; McKeon et al. 2008). The average height of the tree strip was measured at each of the eight pasture survey sites using a laser rangefinder within the bounds of the bird survey plot. We used these height measurements to express distance from the edge of the strip in terms of average tree height.

From the edge of the strip, the transect ran 20 x H (where H = average height of trees in the strip) into the adjacent pasture. For example, a tree strip 6 m high had the transect run 120 m (20 x 6) into the adjacent pasture from the strip edge. Transects were positioned so that they would intersect the bird survey plot if they continued to run further into the strip (Figure 3).

We chose to survey to 20 x H from the edge of the strip since, in their summary of the Australian National Windbreak research program, Cleugh et al. (2002) described a zone of unchanged or slightly increased yield that extended downwind to 10 H or 20 H. Other studies have demonstrated that the beneficial effects of tree strips on crop and pasture yields have largely diminished by 15 H (Iwasaki et al. 2021; Liu et al. 2022). Furthermore, in their study on tree strip effects on pasture in the BBR, McKeon et al. (2008) sampled to 8 H, and did not find an asymptote in pasture yield, suggesting the tree strip was still influencing pasture yield at 8 H. Hence, we assumed sampling 20 H out from the edge of the strip would allow us to sample the area where a beneficial zone may exist across all strips, in addition to a zone where the strip effect on pasture should be greatly diminished.

Benefits derived from tree strips are most common on the leeward side of strips (Cleugh et al. 2002). However, we were unable to confidently determine a prevailing wind direction. Regardless, in most locations, we were only able to survey on one side of most strips, since access tracks and fencelines often ran parallel along the edges of strips, affecting the ability to conduct pasture surveys on both sides of the strip.

Pasture basal area measurements

We used pasture basal area as a proxy for pasture yield. This is because we did not have the ability to exclude cattle from our transects, which were located in grazed pastures. Hence, pasture biomass estimations may have been affected by uneven grazing along the transect. Preliminary data analysis indicated pasture basal area was a significant positive predictor of pasture biomass in buffel grass dominated pastures (Supplementary Information- Figure S1). For this preliminary analysis, we cut and bagged pasture from a random selection of 0.5 x 0.5 m gridded quadrats that were used to measure pasture basal area. The bagged pasture was then dried for three days at 65°C, and its weight measured to determine the correlation between pasture basal area (%) and pasture dry matter yield (tonnes/ha).

Gridded quadrats were used to sample pasture basal area along a central transect. We sampled at 2 m intervals from the strip edge (0 H) to 10 H, and at 5 m intervals from 10–20 H. Electric shears were used to cut pasture to a height of 15 cm. Once the pasture was cut, a gridded 0.5 x 0.5 m quadrat was placed over the cut pasture, and the surveyor (JP) estimated the percentage of pasture basal area within the quadrat.

Pasture basal area readings from the same distances along the three transects were then averaged, so each data point represented the average of three measurements.

Pasture data analysis

All pasture data analysis was completed using Genstat (Genstat 2022).

We used a nonlinear regression analysis to model the change in pasture basal area as a function of the distance from the strip edge, following an approach used by Bennel and Verbyla (2008). Based on previous studies, we expected pasture basal area to respond to distance from the strip edge in a manner similar to Figure 4 (Bennell & Verbyla 2008; Cleugh et al. 2002; Iwasaki et al. 2021; Liu et al. 2022).

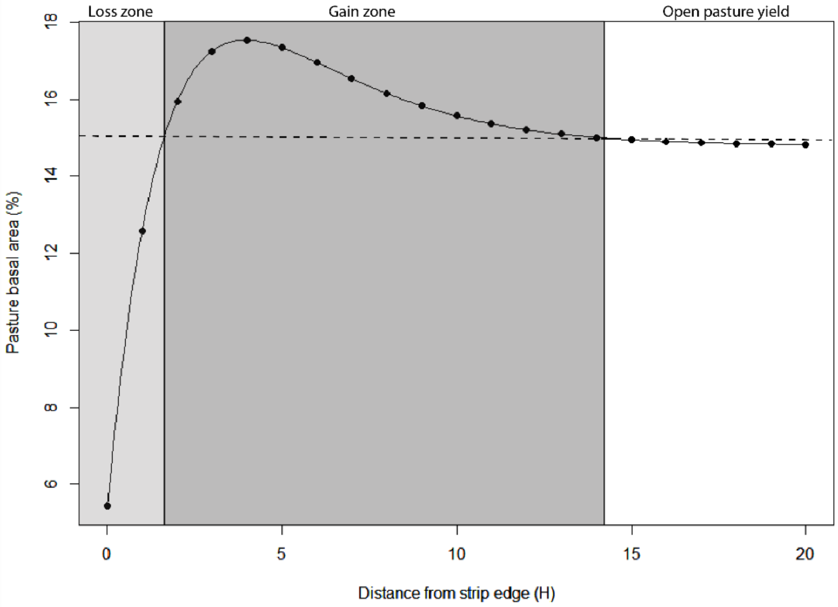


Figure 4 The expected relationship between distance from the strip edge and pasture basal area, and the three expected zones of production. The curve is described by: $f(\mathrm{x}) = A + (B + C\mathrm{x}) \times R^{\mathrm{x}}$, where x is the distance from the strip edge, expressed as a multiple of average tree height (H). *A, B, C* and *R* are the fitted coefficients. The dotted line indicates the 'open pasture yield', when the fitted curve is within 1% of the asymptote (*A*). Figure design modified from Bennell & Verbyla 2008, pg. 952.

We used the model terms (*A, B* and *C*) to define three key zones of production (Figure 4).

1. A 'zone of open pasture yield', where we assumed the effects of the tree strip on pasture basal area have reduced to minute amounts. This zone began when the fitted curve came within 1% of the asymptote (*A*) as the distance from the strip increased (Bennell & Verbyla 2008). We were unable to establish a 'true' control of open pasture that would have no effect of tree strips since each property's soil type and landscape form varied considerably.
2. A 'zone of loss', where pasture basal area ranged below the 'open pasture yield'. This zone was defined as starting at the edge of the tree strip and finishing when x= *-B/C*.
3. A 'zone of gain', where pasture basal area ranged above the 'open pasture yield'. This zone was bounded by the loss zone and the open pasture zone.

RESULTS

We detected 57 different land bird species in sites within the strips, of which 35 were classified as woodland birds. A full species list is provided in the supplementary information (Table S1). Woodland birds were present at all survey sites, with richness ranging from 1 to 13 (mean = 5.5 species). Nine of the 35 woodland birds were detected at more than 10 sites.

The five explanatory variables also varied considerably between sites. 18 sites had *Manorina* honeyeaters, with counts ranging from 0 to 13 birds per site (mean 1.9). Of the 47 sites surveyed, 13 were classed as regrowth and 34 remnant. Strip width ranged from 30 to 388 m (mean 104 m), and buffel grass was absent at only two sites, with cover ranging from 0 to 16.1% (mean 4.2%). Finally, remnant and high value regrowth cover within 1 km ranged from 0 to 47.3% (mean 16.8%).

The relative importance of strip characteristics for woodland birds

Woodland bird richness

There were six models within two AICc values of the best-performing model of woodland bird richness. *Manorina* spp. abundance was included in all six models. The null model was not present within four AICc values of the best model (See supplementary information for summary of all models within four delta values of the best model- Table S2). The global model explained 38% of the variance in woodland bird species richness, based on marginal R^2.

The abundance of *Manorina* spp. was the only important predictor of woodland bird richness at sites (Figure 5a). Fewer bird species were detected in sites that had greater levels of *Manorina* spp. abundance. Neither width, vegetation status, buffel grass cover or vegetation cover was important in explaining woodland bird species richness in the study area.

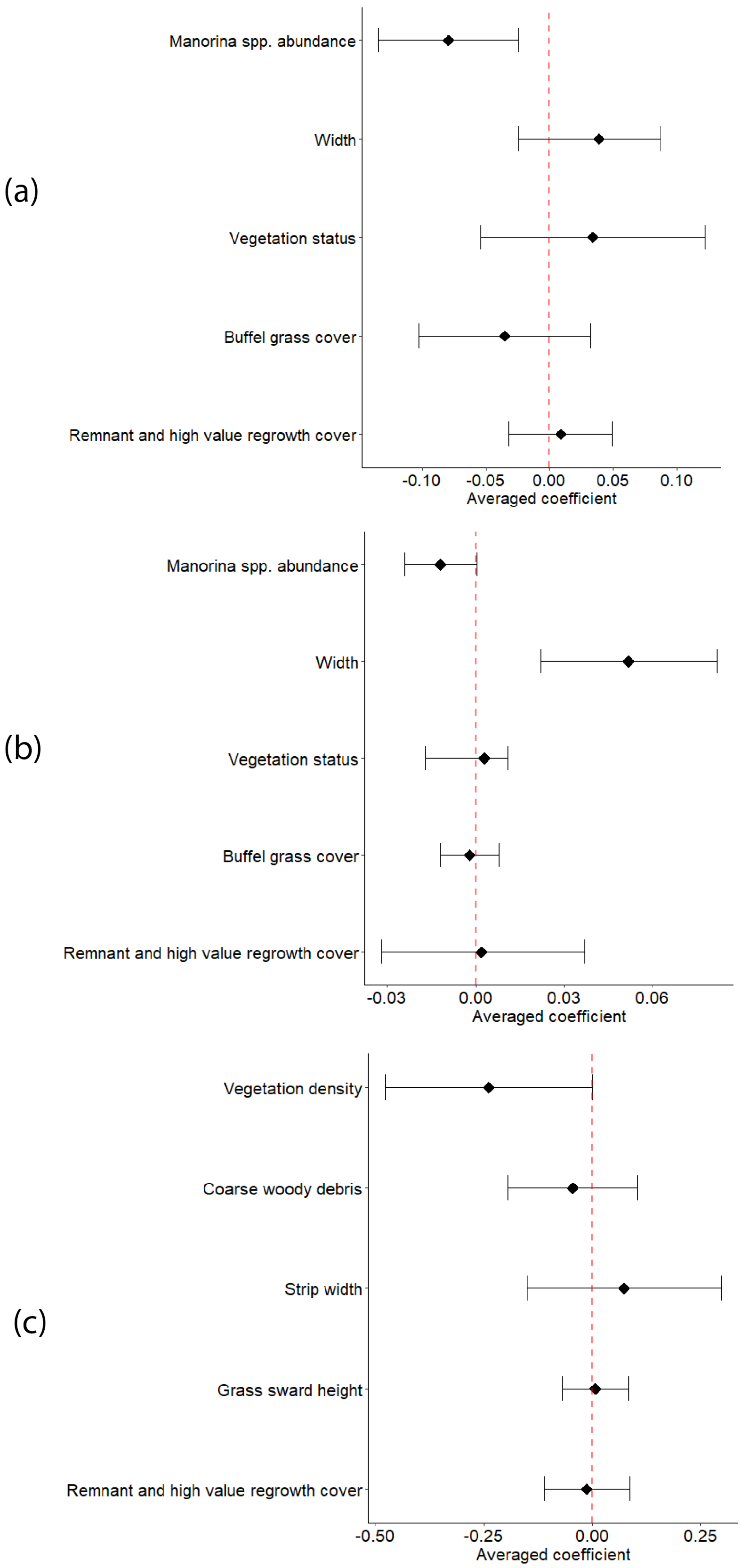


Figure 5- Averaged coefficients and 95% confidence intervals for **(a)** woodland bird richness **(b)** woodland bird abundance **(c)** *Manorina* spp. abundance. Diamond symbols are the averaged coefficients and error bars are the 95% confidence intervals.

Woodland bird abundance

There were three models within two AICc values of the best model of woodland bird abundance. Both *Manorina* spp. abundance and strip width were included in these three models. The null model was not present within four AICc values of the best model (see supplementary information Table S2). The global model explained 48% of the variance in woodland bird species abundance.

Strip width was the only important predictor of woodland bird abundance (Figure 5b). More woodland bird species were detected in sites in wider strips. Neither *Manorina* spp. abundance, vegetation status, buffel grass cover or vegetation cover was important in explaining woodland bird abundance.

Manorina *spp. abundance*

Four models were within 2 AICc values of the best model of *Manorina* spp. abundance, indicating moderate model uncertainty, and all included vegetation density as an explanatory variable. The null model was not present within 4 AICc of the best model.

No explanatory variables had a significant relationship with *Manorina* spp. abundance (Figure 5c). The global model explained 30% of the variance in *Manorina* spp. abundance.

Strip width effects on habitat characteristics

Strip width was significantly positively associated with vegetation structural complexity and large trees (Figure 6). A linear function provided the best fit for vegetation structural complexity, whilst a log function was most suitable for large trees. The AICc weights of each model of the fitted models are provided in the supplementary information (Table S3). Strip width was not significantly associated with mistletoe abundance, buffel grass cover, coarse woody debris or vegetation density.

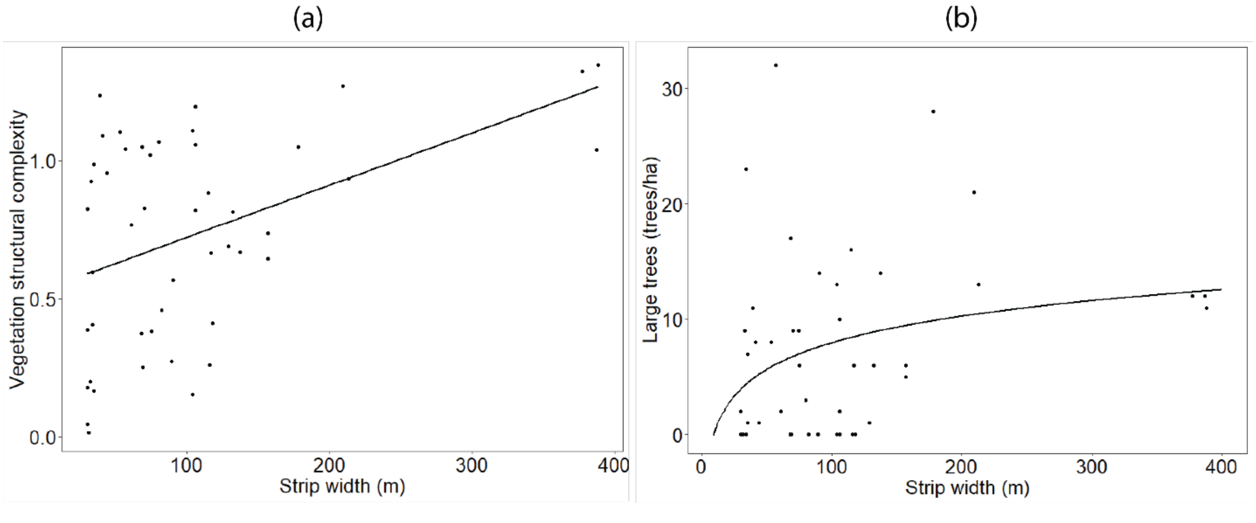


Figure 6 The response of ***(a)*** vegetation structural complexity and ***(b)*** large trees to strip width. Each data point represents an individual survey site (n =47).

The effect of tree strips on adjacent pasture basal area

We generated nonlinear regression models of the form $f(\mathrm{x}) = A + (B + C\mathrm{x}) \times R^{\mathrm{x}}$ for each site, between the distance from the strip edge and pasture basal area (Table 2 and Figure 7). Initial models showed the *R* term was not significantly different across sites ($F_{7,356}$ = 0.76; P = 0.62), and thus a pooled coefficient for *R* was adopted (*R* = 0.66). *A, B* and *C* did show site differences – *A* ($F_{7,363}$ = 23.0; P <0.001), and *B* and *C* (pooled $F_{14,363}$ = 3.05; P < 0.001).

Table 2 Model parameter values for each site's exponential curve

Parameter	Site ID							
	1	2	3	4	5	6	7	8
A	9.27	18.61	16.21	7.68	14.80	19.55	16.29	16.32
B	3.02	-8.76	-6.91	0.62	-9.37	-16.43	-12.36	-14.93
C	6.43	11.01	3.03	4.42	11.14	3.35	7.88	5.21

There was substantial variation among the eight sites in both the magnitude and the size of the zone of gain (Figure 7 and Table 2). For three sites only a very marginal increase in pasture basal area adjacent to the strip was detected (Figure 7).

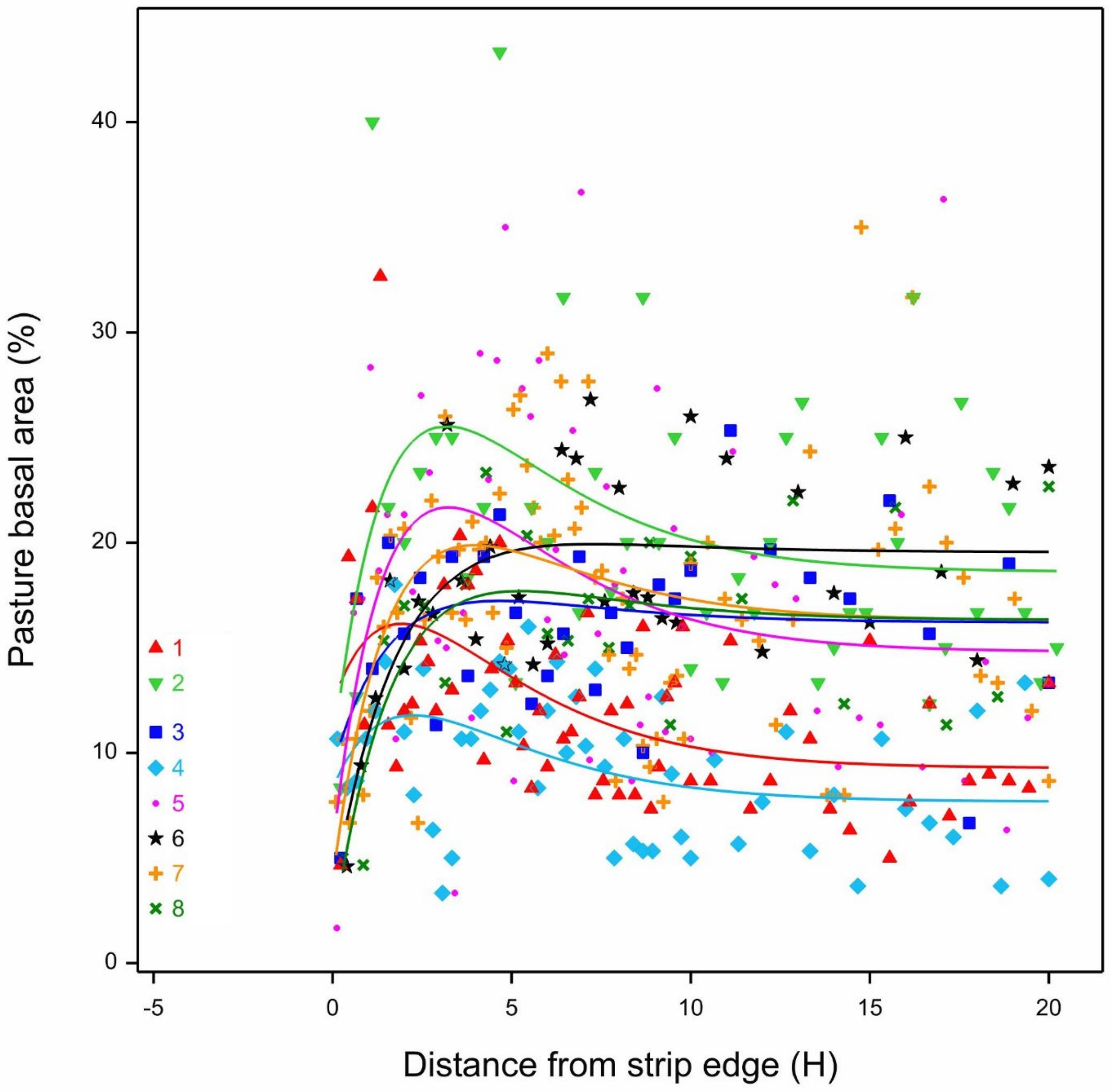


Figure 7 The response of pasture basal area to the distance from the edge of the tree strip, expressed in multiples of the strip's average tree height (H). Labels 1,2,3 etc. represent each of the eight pasture survey sites. Data points represent an average of the multiple transects run at each site. The plotted lines represent each site's exponential model.

Five sites showed substantial gains in pasture basal area that began between 0 – 1.6 H, and finished between 15.5 – 17 H (Table 3).

The size of the zone of loss also varied between sites. Two sites showed no evidence of a loss zone, with the pasture basal area at the edge of the strip being higher than the 'open pasture basal area'. All other sites exhibited loss zones that started at the edge of the strip, and ran to a range of 0.8 – 4.9 H. One site did not exhibit any meaningful beneficial effect, and instead had the largest loss zone from 0 to 4.9 H (Table 3).

Table 3 Analysis of each site's critical exponential model showing variation in 'open pasture basal area' and the loss and gain zones.

Site ID	Open pasture basal area (%)	Loss Zone boundaries (H)	Gain Zone boundaries (H)
1	9	No comp. zone	0 - 17
2	19	0 - 0.8	0.8 - 16
3	16	0 - 2.3	2.3 - 12.6
4	8	No comp. zone	0 - 16.5
5	15	0 - 0.8	0.8 - 17
6	20	0 - 4.9	4.9 - 11.16*
7	16	0 - 1.6	1.6 - 15.5
8	16	0 - 2.9	2.9 - 14.1

*Site 6 has a barely discernible gain zone.

DISCUSSION

We found that brigalow-dominated strips of vegetation support many species of woodland birds in a grazing landscape. Relatively thin strips (<50 m wide) were occupied by a similarly rich assemblage of woodland birds as wider strips. However, sites in wider strips had higher abundances of woodland birds, and the richness of woodland birds was negatively correlated with the abundance of aggressive native birds of the genus *Manorina*. A distinct zone of increased pasture basal area adjacent to strips was detected at five of eight pasture survey sites. These results suggest that the retention of both remnant and regrowth brigalow-dominated vegetation in the form of strips could deliver conservation outcomes for woodland birds, whilst minimising pasture productivity losses.

Linear strips support woodland birds

Strips of brigalow-dominated vegetation in grazing landscapes were used by a range of woodland bird species, including species of conservation concern that are declining elsewhere in eastern Australia (Ford et al. 2001; Woinarski et al. 2006). The importance of linear features as habitat and, potentially, conduit for movement has been identified for other birds in otherwise cleared agricultural landscapes (Bonifacio et al. 2011; Hinsley & Bellamy 2000; Šálek et al. 2022; Saunders & Hobbs 1991). For example, hedgerows and vegetated field margins in the lowland farming landscapes of Britain allow birds to persist in these modified environments (Hinsley & Bellamy 2000). In south-eastern Queensland, Bentley and Catterall (1997) found that corridors and linear remnants supported similar densities of woodland birds compared to larger patches of woodland, and highlighted the importance of these linear features to local and migratory bird populations. Similar to Bowen et al. (2009), who surveyed the woodland birds of the eastern BBR, we found that 'decliner' species from New South Wales and the BBR (e.g. Reid 1999, Woinarksi et al. 2006) were able to persist in brigalow-dominated linear fragments. These included species such as the speckled warbler *Chthonicola sagittata*, grey-crowned babbler *Pomatostomus temporalis*, and varied sitella *Daphoenositta chrysoptera* – all species listed as threatened elsewhere in Australia's woodland zone (*Biodiversity Conservation Act 2016)*.

Within the strips, we detected woodland birds from a variety of different foraging guilds, including nectarivores, insectivores, carnivores, and granivores. Of these, insectivores, particularly those that foraged in the canopy, were most abundant, followed by nectarivores. Strips provided key foraging resources to these guilds. For example, various species of mistletoe, a nectar and fruit bearing hemiparasite, was flowering at 17 of our 47 sites and present at 30. Canopy insectivores and nectarivores are declining in the BBR, largely because land clearing reduces the availability of these key foraging resources (Woinarski et al. 2006). Our results indicate that strips serve to retain some of these critical resources within the agricultural landscape. This agrees with the findings presented by Bowen et al. (2008), who found that thin linear strips of remnant vegetation supported greater abundances of mistletoe than less-disturbed, larger patches, and where therefore probably important to maintaining populations of nectarivorous birds throughout the region.

Retained strips in these agricultural landscapes are important contributors to habitat cover, potentially keeping it above critical threshold levels below which woodland bird richness rapidly declines (Maron et al. 2012; Radford et al. 2005; Simmonds et al. 2019). In northern Victoria, Radford et al. (2005) found that in agricultural landscapes with habitat cover <10%, woodland bird richness rapidly declined. Within the BBR, Maron et al. (2012) also found evidence of threshold habitat cover values, but that these threshold values could be mediated by the landscape's productivity. For example, species richness increased steeply with increasing habitat area in high productivity landscapes compared to low-productivity landscapes (Maron et al. 2012). As our survey sites were in brigalow-dominated vegetation, a vegetation type that grows on higher-productivity lands (Seabrook et al. 2006), retained strips of brigalow-dominated vegetation may have a greater impact on the landscape's woodland bird richness than larger areas of lower-productivity vegetation. However, Simmonds et al. (2019) showed that the 'tipping point' at which these thresholds occur vary widely across eastern Australia. This can make the 'location' of thresholds difficult to determine and an unreliable measure upon which to base conservation targets.

Finally, strips could improve connectivity across the study area, and may be used by birds for dispersal and to access resources that are risky or otherwise hard to get to (Bentley & Catterall 1997; Haas 1995; Hinsley & Bellamy 2000). For example, Saunders and De Rebeira (1991) demonstrated that linear strips were important for woodland bird dispersal in the highly-modified West Australian Wheatbelt.

It is important to note that some woodland bird species that use brigalow-dominated vegetation, and whose ranges overlap with our survey area, were not detected in any of our sites. For example, we did not detect any white-throated treecreepers *Cormobates leucophaeus*. These are a species whose dispersal has been shown to be sensitive to habitat fragmentation (Robertson & Radford 2009). Hence, the lack of retained brigalow-dominated vegetation that forms patches of sufficient size to support fragmentation-sensitive species like the white-throated treecreeper in our study landscape may mean that they no longer occur within it. Instead, the woodland birds present within strips may represent a 'strip tolerant' assemblage, from which some area-sensitive species are missing. Further research could test whether larger patches of remnant brigalow in this region have different woodland bird species assemblages, which would indicate whether strips provide habitat resources for only a subset of woodland birds.

Strip width affects woodland bird abundance but not richness

We found that strip width was not an important predictor of woodland bird species richness at a site. Sites in relatively thin strips (< 50 m) supported a similar species richness to wider strips (>50 m), with thin strips supporting an average woodland bird richness of 4.6 ± 2.3 SD, compared to the average richness of 5.9 ± 3 SD that wider strips supported. This was contrary to our expectations, given that the literature has often reported that the width of linear strips is positively associated with site-level bird species richness (Bonifacio et al. 2011; Hall et al. 2018; Hinsley & Bellamy 2000). For example, Hall et al. (2018) found that the width of riparian and roadside linear strips was significantly positively correlated with the richness of woodland birds within them. Edge effects may not have been particularly important for the woodland bird assemblage that we examined, even though narrow strips that were surveyed naturally contained predominantly edge habitat compared to wider strips, in which we surveyed more 'core habitat'. However, most studies on Australian woodland birds have failed to detect any significant edge effects (Baker, French & Whelan 2002; Berry 2001; Campi & MacNally 2001), except in highly modified urban environments (e.g. Catterall et al. 1992). So although thinner strips were composed of more edge habitat, edge-tolerance of the woodland birds within our sites may serve to reduce the effect of strip width on species richness.

Although the richness of woodland birds in a site was not affected by strip width, woodland bird abundance was positively correlated with strip width. Wider strips tended to have greater vegetation structural complexity (Figure 6). Greater structural complexity has often been associated with greater bird abundance (Eyre et al. 2009; Maron & Kennedy 2007; Martin & Possingham 2005). For example, Eyre et al. (2009) showed that midstory stem density had an important positive effect on the abundance of small woodland birds. Maron and Kennedy (2007) found that sites with less understory supported lower densities of woodland birds, but that this effect may be driven by the impacts of the vegetation complexity on *Manorina* spp., which tend to be less abundant in sites with well-developed understory (Howes & Maron 2009; Lindenmayer et al. 2010). Sites with greater vegetation structural complexity also likely provide more habitat resources per unit area than those with a more simplified structure (Lindenmayer et al. 2018; Montague-Drake et al. 2009), in turn supporting a greater abundance of woodland birds (Martin & Possingham 2005).

Lower vegetation structural complexity in thinner strips may be due to these strips suffering from increased grazing pressure, since cattle have less area to spread out whilst sheltering and grazing within strips. For example, increased grazing pressure has been shown to simplify the structural complexity of sites through the removal of shrub and understory layers, leading to negative impacts on woodland bird abundance (Martin & McIntyre 2007).

Narrow strips may also be more subject to edge effects, like increased wind speed, air temperature, light intensity, and invasion by exotic species, that could drive changes in vegetation structural complexity (Murcia 1995). Whilst we found no significant relationship between strip width and buffel grass cover (Table S3), other work has shown brigalow woodland edges can be prone to invasion by buffel grass (Butler & Fairfax 2003). Buffel grass invasion into brigalow woodland increases ground fuel loads, leading to increased fire intensity and frequency that simplifies vegetation structural complexity (Butler & Fairfax 2003). Further investigation into the interaction between strip width and vegetation structural complexity is warranted, so possible deleterious edge effects may be managed within strips.

Large trees were also more common in wider strips (Figure 6), and this may also contribute to the positive effect of strip width on woodland bird abundance. In other studies in eastern Australia, the presence of large trees was an important driver of bird breeding activity in both revegetation (Selwood, Mac Nally & Thomson 2009) and remnant sites (Bennett et al. 2015), which may serve to increase bird abundance. Large trees provide nesting hollows that are required by some woodland birds for breeding (Bennett 1994), are an important source of nectar, support abundant invertebrate assemblages, and have large canopies for foraging and nesting (Vesk et al. 2008). It is reasonable to expect that the greater chances of these important resources being provided in wider strips might contribute to strip width's positive effect on abundance.

A limitation of this study was that we were not able to directly test the effect of other explanatory variables like large trees on woodland bird richness and abundance. Due to short time frames and difficulties identifying suitable survey sites within a small geographic area so as not to introduce substantial environmental variations, our sample size of 47 sites meant that we risked overfitting if we included too many explanatory variables within our GLMMs. Expanding the number of survey sites would allow greater insights on the effects of a larger range of strip characteristics on woodland birds.

Vegetation status and surrounding habitat cover do not affect woodland bird richness and abundance

Other studies in the BBR and beyond have found that the vegetation age of revegetation or regrowth was positively correlated with woodland bird richness (Bowen et al. 2009; Haslem et al. 2021), but we did not find this. This difference may be because several sites in our study that are classified as 'remnant' based upon clearing history as recounted by landowners have resprouted following hot fires, resulting in suckering and growth patterns similar to regrowth stands. Fire degradation has been proposed as a mechanism that allows *Manorina* honeyeater range expansion and resultant declines in small woodland bird richness and abundance (Maron & Kennedy 2007). This may have served to diminish the differences in woodland bird abundance and richness between remnant and regrowth sites.

Further, regrowth sites of all ages were grouped into the one class to achieve sufficient sample size, meaning we could not differentiate patterns between younger and older regrowth. Older regrowth may support greater woodland bird richness, since Bowen et al. (2009) found that the greater the amount of older regrowth vegetation (>30 yrs) surrounding a site, the greater the woodland bird abundance compared to sites surrounded by younger regrowth. So whilst our results suggest regrowth supports similar woodland bird richness and abundance as remnant vegetation, surveying the bird populations of regrowth strips of a variety of different ages rather than using broad definitions of 'remnant' and 'regrowth' would yield important information regarding potential successional changes in woodland bird communities over time. This would provide information regarding expected benefits of regrowth retention to woodland birds, which is especially important since any increases in the number and extent of strips throughout the region will rely on strategic retention of brigalow regrowth.

The amount of remnant and high value regrowth vegetation in the landscape surrounding each survey site did not affect woodland bird richness and abundance. Other studies in Australian agricultural landscapes found that surrounding habitat cover was an important influence on woodland bird communities, both at the patch (Bowen et al. 2009) and landscape scales (Bennett et al. 2022; Radford et al. 2005). Our study investigated surrounding habitat cover at the patch scale, but compared to Bowen et al. (2009), who sampled surrounding remnant vegetation cover from 0% to 100%, we were only able to sample a relatively small range of vegetation cover (0% to 47%, including both remnant and regrowth vegetation). We therefore may not have sampled enough variation in vegetation cover to detect a significant effect. However, finding strips of brigalow-dominated vegetation with high surrounding habitat cover in 1 km radius was not possible within this landscape.

Aggressive native honeyeaters and their impact on woodland birds

The collective abundance of the two *Manorina* spp. had a negative effect on the richness of woodland birds within our survey sites. This result accords with those of many other studies in the woodlands of eastern Australia, where *Manorina* spp. are abundant (Bowen et al. 2009; Eyre et al. 2009; Hall et al. 2018; Maron & Kennedy 2007; Thomson et al. 2015). The impact of native birds in this genus on other species is widely recognised in Australia, and the phenomenon is listed as a Key Threatening Process under Australia's EPBC Act (1999). *Manorina* spp. actively defend their territories, and physically exclude small-bodied passerine birds from their home ranges (Mac Nally et al. 2012; Maron 2009).

Whilst *Manorina* spp. abundance had a negative effect on woodland bird richness, it had no significant effect on abundance. We suggest that this may be because of different responses of woodland birds to increased *Manorina* spp. abundance. For example, Howes et al. (2014) found that as *Manorina* spp. abundance increased, sites were increasingly dominated by larger woodland birds, including large insectivores, granivores and carnivores, whilst small insectivores and frugivores significantly decreased. Hence, an increase in the number of larger woodland birds may have offset losses in abundance of smaller woodland birds, resulting in *Manorina* spp. abundance not having a significant effect on overall woodland bird abundance.

We were unable to explain the variation in *Manorina* spp. abundance among our survey sites, with none of the five explanatory variables significantly correlated to *Manorina* spp. abundance. This result was unexpected, since many studies have found the density of understory and midstory vegetation and grazing intensity to be important drivers of *Manorina* spp. abundance (Eyre et al. 2009; Hastings & Beattie 2006; MacDonald & Kirkpatrick 2003; Maron & Kennedy 2007). For example, Eyre et al. (2009) found that a site's midstory count, grazing intensity and coarse woody debris (measured as vegetation density, average pasture height and coarse woody debris respectively in this study) were significant predictors of *Manorina* spp. abundance. Further research that aims to determine what strip characteristics are driving *Manorina* spp. abundance would allow landholders to potentially manage strips in a way that makes them less suitable for *Manorina* spp. colonisation.

Since the drivers of *Manorina* spp. abundance within our study landscape are unclear, if future conservation strategies aim to reduce the impact of *Manorina* on woodland birds, other strategies may need to be considered. These include direct management strategies; specifically, the culling of *Manorina* honeyeaters to reduce abundance (Crates et al. 2018). Whilst considerable uncertainty surrounds the efficacy of culls for long-term woodland bird benefit, short- and medium-term gains in woodland bird richness following culling events suggest that strategic and targeting culling may result in improved outcomes for woodland birds (Melton et al. 2021). However, we did not detect any woodland birds of very high conservation concern (e.g. those listed as Vulnerable or above on the EPBC Act (1999)) within our sites, so potentially controversial interventions like culling may not easily justified (Melton et al. 2021).

Strips affect adjacent pasture yields

As observed in previous studies in cropping systems (Bennell & Verbyla 2008; Liu et al. 2022; Osorio, Barden & Ciampitti 2019), strips of woody vegetation appeared to be associated with three pasture production zones at most of our sites. These were a zone of open pasture yield where pasture basal area stabilised beyond the effect of the strip, a loss zone where pasture basal area was below that of the 'open pasture yield'; and a gain zone, where pasture basal area was higher than in 'open pasture'. The loss zone started at the edge of the strip and extended up to 4.9 H into the adjacent pasture. The gain zone began between 0 – 2.9 H, and extented to a maximum of 17 H. This is similar to the results of Bennell and Verbyla (2008), who found that their beneficial zone extented to a maximum of 20 H in cereal grain croplands.

Multiple studies have proposed mechanisms by which strips affect adjacent pasture yield. For example, changes to microclimate in the lee of strips has been cited as the main driver of the 'zone of gain' in another study investigating the effect of tree strips on pasture yield in the BBR (McKeon et al. 2008). Strips may modify the local temperature, relative humidity, evapotranspiration and windspeed in their lee, reducing stressors on pastures and crops (Brenner 1996; Cleugh et al. 2002). A reduction in evaporative losses from soil can increase plant available soil water and allow easier water absorption as rewetting is not as frequently required (Bird 1998; Ryan et al. 2010). In the cropping regions of Australia, the largest yield gains were simulated where the latter part of the growing season is characteristed by high evaporation and depleted soil moisture, which can occur in our study region (Cleugh et al. 2002). Further investigation that explains what factors are driving the beneficial effect in our study context, and how these factors relate to strip characteristics will allow land owners to manage strips for maximum potential benefit to pasture.

The formation of a 'loss zone' was probably driven by competition for light, water and nutrients between the brigalow-dominated vegetation and buffel pasture, resulting in poorer pasture productivity (Scanlan 2002). Being adapted for semi-arid environments, brigalow trees have well-developed lateral roots in the upper 30 cm of the soil profile, causing root-zone overlap with pasture (Johnson 1964). Most soil-water flucations occur within the top 1 m of the soil profile underneath mature brigalow trees, highlighting the potential for tree-pasture competion (Tunstall & Connor 1981). As such, increases in the basal area of brigalow trees has been shown to negatively correspond with pasture basal area both below the brigalow canopy, and immediately beside brigalow stands (Scanlan 1991).

Implications for conservation on private agricultural land

We found that strips of brigalow-dominated vegetation were used by declining woodland bird communities, and there was strong evidence of a positive pasture productivity effect adjacent to strips. This positive effect could help to offset some of the losses in pasture productivity that occur directly beneath the tree strip and in the loss zone, reducing the overall costs of strip retention to land managers.

We suggest that the widening and preservation of existing strips, and the creation of new strips from regrowth, could be effective methods that improve the woodland bird population on a property and contribute to the restoration of endangered brigalow ecosystems in the region. These strategies would increase the total extent of wooded vegetation and the diversity in succession growth stages, providing a wide variety of habitat resources (Bennett et al. 2022). To test whether these strategies will improve the conservation status of declining woodland birds at the landscape scale, we suggest further research to compare woodland bird communities among replicate landscapes that vary in the number and width of retained strips and *Manorina* spp. abundances, following a similar approach to Bennett et al. (2022). This would reveal insights on the effect of strip retention on the broader land mosaic and how this influences woodland bird populations.

It will also be important to determine what extent, quality and configuration of habitat within grazing landscapes would be required to recover fragmentation-sensitive species like the white-throated treecreeper, that we did not detect within any sites. Recovering these species will likely require the regeneration of larger, more continuous tracts of vegetation that may incur greater costs on agricultural production (Cooper et al. 2002; Robertson & Radford 2009). Determining the costs of such restoration (Mappin et al. 2022), identifying areas of high-restoration priority (Bennett & Mac Nally 2004; Crossman & Bryan 2009), and evaluating whether current market-based mechanisms can offset these increased costs to agricultural businesses (Donaghy et al. 2010), will be necessary to recover the full suite of woodland birds that formerly occupied the region.

From an agricultural perspective, increases in the number and width of strips may result in net pasture losses if the beneficial zone cannot compensate for losses both directly within the strip, and in the loss zone immediately adjacent. It is likely that a strip width and/or density would be reached at which pasture losses directly beneath the strip outweigh gains made from the gain zone in the adjacent pasture (McKeon et al. 2008). These losses must be known so that graziers can make informed decisions regarding the trade-offs between lost pasture production, and increased yields from the gain zone. Such information is critical when attempting to encourage widespread uptake of conservaton measures on private agricultural land (Pannell et al. 2006).

The retention of more and wider strips may become more economically feasible if graziers are able to access financial benefits from environmental markets and/or biodiversity certifications, which would help to offset potential pasture productivity losses and provide a diversification of income. We have demonstrated the strips are used by declining woodland birds on private grazing land, and this should be considered within such schemes. Other research has demonstrated that brigalow-eucalypt strips retained from regrowth also store substantial above-ground biomass (Ryan et al. 2015). This means that land managers who retain strips of brigalow regrowth on their property that otherwise would have been cleared may receive recognition of the biodiversity and carbon benefits provided by the retained vegetation, allowing them to sell 'biodiversity certificates' and access premium and carbon markets (Gibson 2022; Gowen & Bray 2016). This is especially relevant as governments and NGOs look to privately owned land to improve conservation and carbon sequestration outcomes in the BBR (Department of Climate Change, Energy, Environment and Water 2022; Queensland Trust for Nature n.d.).

If benefits to landholders from environmental markets and certifications are unable to offset costs associated with potential productivity losses incurred by the retention of wider strips, thinner strips would be more likely favoured by land managers since they will probably still provide shade and shelter benefits, but will occupy less total land area. Our results indicate that a similar richness of woodland birds will use relatively thin strips compared to those that are wider, albeit at lower abundances. Therefore, even if the retention of wider strips is considered economically unfeasible and unlikely to be adopted by an enterprise, the retention of thinner strips could still deliver important habitat for a range of woodland birds.

CONCLUSIONS

To ensure widespread uptake and adoption, conservation strategies on private land should minimise impacts on profitable and productive businesses. Retained brigalow-dominated strips can provide important habitat resources for woodland birds whilst minimising pasture losses. Even narrow strips were used by a range of woodland birds, indicating their importance for woodland bird conservation in agricultural landscapes. The abundance of hyper-aggressive honeyeaters was negatively correlated with the abundance and richness of woodland birds, whilst strip width was positively associated with woodland bird abundance. Strips can drive both pasture losses and gains in adjacent paddocks. We propose that the retention of new strips, and the widening and preservation of existing strips, should be considered as valid methods for land managers seeking to enter environmental markets and improve their on-farm biodiversity in a way that minimizes costs to overall production.

ACKNOWLEDGEMENTS

We thank the A.W. Howard Memorial Trust, who generously provided an Honours scholarship for this project. Thanks to the private land holders who welcomed us onto their properties with hospitality and provided much appreciated accommodation. Thanks to Hayley McMillan, Bradd Witt and Martine Maron whose comments and guidance improved this thesis. Joshua Peart was an incredible help with completing the field work. Our research was conducted on the traditional lands of the Bidjara people. We acknowledge their Elders past, present and future.

REFERENCES

Accad, A, Kelley, JAR, Richter, D, Neldner, VJ & Li, J, 2022, *Remnant Regional Ecosystem Vegetation in Queensland (Version 12.2), Analysis 1997 - 2019*, Department of Environment and Science, State of Queensland, Brisbane.

Baker, J, French, K & Whelan, RJ 2002, 'The Edge Effect and Ecotonal Species: Bird Communities across a Natural Edge in Southeastern Australia', *Ecology (Durham)*, vol. 83, no. 11, pp. 3048-59.

Bates D, Mächler M, Bolker B, Walker S 2015. 'Fitting Linear Mixed-Effects Models Using lme4.' Journal of Statistical Software, 67(1), 1–48.

Barth, BJ, Fitzgibbon, SI, Gillett, A, Wilson, RS, Moffitt, B, Pye, GW, Adam, D, Preece, H & Ellis, WA 2020, 'Scattered paddock trees and roadside vegetation can provide important habitat for koalas (Phascolarctos cinereus) in an agricultural landscape', *Australian mammalogy*, vol. 42, no. 2, pp. 194-203.

Barton, K 2022 'MuMIn: Multi-Model Inference'. R package version 1.47.1, <https://CRAN.R-project.org/package=MuMIn>.

Bennell, MR & Verbyla, AP 2008, 'Quantifying the response of crops to shelter in the agricultural regions of South Australia', *Australian journal of agricultural research*, vol. 59, no. 10, pp. 950-7.

Bennett, AF & Mac Nally, R 2004, 'Identifying priority areas for conservation action in agricultural landscapes', *Pacific conservation biology*, vol. 10, no. 2, pp. 106-23.

Bennett, AF 1994, 'Tree hollows as a resource for wildlife in remnant woodlands: spatial and temporal patterns across the northern plains of Victoria, Australia', *Pacific conservation biology*, vol. 1, no. 3, pp. 222-.

Bennett, AF, Holland, GJ, Haslem, A, Stewart, A, Radford, JQ & Clarke, RH 2022, 'Restoration promotes recovery of woodland birds in agricultural environments: A comparison of ‘revegetation’ and ‘remnant’ landscapes', *The Journal of applied ecology*, vol. 59, no. 5, pp. 1334-46.

Bennett, JM, Clarke, RH, Thomson, JR & Mac Nally, R 2015, 'Fragmentation, vegetation change and irruptive competitors affect recruitment of woodland birds', *Ecography*, vol. 38, no. 2, pp. 163-71.

Bentley, JM & Catterall, CP 1997, 'The Use of Bushland, Corridors, and Linear Remnants by Birds in Southeastern Queensland, Australia', *Conservation Biology*, vol. 11, no. 5, pp. 1173-89.

Berry, L 2001, 'Edge effects on the distribution and abundance of birds in a southern Victorian forest', *Wildlife research (East Melbourne)*, vol. 28, no. 3, pp. 239-45.

Bianchi, FJJA, Mikos, V, Brussaard, L, Delbaere, B & Pulleman, MM 2013, 'Opportunities and limitations for functional agrobiodiversity in the European context', *Environmental science & policy*, vol. 27, pp. 223-31.

Biodiversity Conservation Act 2016 (NSW)

Bird, PR 1998, 'Tree windbreaks and shelter benefits to pasture in temperate grazing systems', *Agroforestry systems*, vol. 41, no. 1, pp. 35-54.

Bird, PR, Bicknell, D, Bulman, PA, Burke, SJA, Leys, JF, Parker, JN, Van Der Sommen, FJ & Voller, P 1992, 'The role of shelter in Australia for protecting soils, plants and livestock', *Agroforestry systems*, vol. 20, no. 1-2, pp. 59-86.

Bird, PR, Jackson, TT, Kearney, GA & Williams, KW 2002, 'Effect of two tree windbreaks on adjacent pastures in south-western Victoria, Australia', *Australian journal of experimental agriculture*, vol. 42, no. 6, pp. 809-30.

Bureau of Meteorology 2022a, *Climate statistics for Australian locations*, viewed 30/09/2022, <http://www.bom.gov.au/climate/averages/tables/cw_044021.shtml>.

Bureau of Meteorology 2022b, *Monthly Rainfall Augathella Post Office*, viewed 30/09/2022 2022, <http://www.bom.gov.au/jsp/ncc/cdio/weatherData/av?p_nccObsCode=139&p_display_type=dataFile&p_startYear=&p_c=&p_stn_num=044002>.

Bonifacio, RS, Kinross, CM, Gurr, GM & Nicol, H 2011, 'The effect of woody plant diversity and other stand and landscape factors on the diversity and abundance of birds using farm shelterbelts', *Pacific conservation biology*, vol. 17, no. 1, pp. 22-35.

Bowen, ME 2009, 'Quantifying the ecological values of brigalow regrowth for woodland birds: a hierarchical landscape approach', PhD thesis, University of Queensland, Brisbane.

Bowen, ME, McAlpine, CA, House, APN & Smith, GC 2007, 'Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna', *Biological Conservation*, vol. 140, no. 3, pp. 273-96.

Bowen, ME, McAlpine, CA, Seabrook, LM, House, APN & Smith, GC 2009, 'The age and amount of regrowth forest in fragmented brigalow landscapes are both important for woodland dependent birds', *Biological Conservation*, vol. 142, no. 12, pp. 3051-9.

Bradley, M 2007, 'Integration of Brigalow (*Acacia harpophylla*) Vegetation with Dryland Cropping in South-East Queensland: Agronomic, Hydrological, Ecological and Goethean Perspectives', PhD thesis, Griffith University, Brisbane.

Brandle, JR, Hodges, L & Zhou, XH 2004, 'Windbreaks in North American agricultural systems', *Agroforestry systems*, vol. 61, no. 1, pp. 65-78.

Brenner, AJ 1996, 'Microclimate Modifications in Agroforestry', in C Ong & P Huxley (eds), *Tree-Crop Interactions A Physiological Approach*, CAB INTERNATIONAL, Cambridge, vol. 1.

Burnham, KP & Anderson, DR 2002, *Model selection and multi-model inference : a practical information-theoretic approach*

Model selection and multimodel inference : a practical information-theoretic approach, 2nd edn, Springer, New York.

Butler, D & Fairfax, RJ 2003, 'Buffel Grass and fire in a Gidgee and Brigalow woodland: A case study from central Queensland', *Ecological Management & Restoration*, vol. 4, no. 2, pp. 120-5.

Campi, MJ & MacNally, R 2001, 'Birds on edge: avian assemblages along forest-agricultural boundaries of central Victoria, Australia', *Animal conservation*, vol. 4, no. 2, pp. 121-32.

Chazdon, RL, Peres, CA, Dent, D, Sheil, D, Lugo, AE, Lamb, D, Stork, NE & Miller, SE 2009, 'The Potential for Species Conservation in Tropical Secondary Forests', *Conservation Biology*, vol. 23, no. 6, pp. 1406-17.

Clarke, MF & Oldland, JM 2007, 'Penetration of remnant edges by noisy miners (Manorina melanocephala) and implications for habitat restoration', *Wildlife research (East Melbourne)*, vol. 34, no. 4, pp. 253-61.

Cleugh, HA, Prinsley, R, Bird, PR, Brooks, SJ, Carberry, PS, Crawford, MC, Jackson, TT, Meinke, H, Mylius, SJ, Nuberg, IK, Sudmeyer, RA & Wright, AJ 2002, 'The Australian National Windbreaks Program: overview and summary of results', *Australian journal of experimental agriculture*, vol. 42, no. 6, pp. 649-64.

Collard, S, Le Brocque, A & Zammit, C 2008, 'Bird assemblages in fragmented agricultural landscapes: the role of small brigalow remnants and adjoining land uses', *Biodiversity and conservation*, vol. 18, no. 6, pp. 1649-70.

Cooper, CB, Walters, JR & Ford, HA 2002, 'Effects of remnant size and connectivity on the response of Brown Treecreepers to habitat fragmentation', *Emu*, vol. 102, no. 3, pp. 249-56.

Crates, R, Terauds, A, Rayner, L, Stojanovic, D, Heinsohn, R, Wilkie, C & Webb, M 2018, 'Spatially and temporally targeted suppression of despotic noisy miners has conservation benefits for highly mobile and threatened woodland birds', *Biological Conservation*, vol. 227, pp. 343-51.

Crossman, ND & Bryan, BA 2009, 'Identifying cost-effective hotspots for restoring natural capital and enhancing landscape multifunctionality', *Ecological Economics*, vol. 68, no. 3, pp. 654-68.

Department of Climate Change, Energy, Environment and Water 2022, *EPBC Act List of Threatened Ecological Communities*, viewed 15/07/2022, <http://www.environment.gov.au/cgi-bin/sprat/public/publiclookupcommunities.pl>.

Department of Environment and Science 2015, *Wooded extent and foliage projective cover code layer - Queensland 2013*, QldSpatial, State of Queensland, Brisbane, viewed 28/10/2022, <http://qldspatial.information.qld.gov.au/catalogue/custom/search.page?q=%22Wooded extent and foliage projective cover code layer - Queensland 2013%22>.

Department of Environment and Science 2022a, *High value regrowth 2019 - Queensland*, QldSpatial, State of Queensland, Brisbane, viewed 28/10/2022, https://qldspatial.information.qld.gov.au/catalogue/custom/detail.page?fid={AA49DD91-1D59-40C1-B9C7-BC472148B90E}

Department of Environment and Science 2022b, *Remnant vegetation cover - 2019 - Queensland*, QldSpatial, State of Queensland, Brisbane, viewed 28/10/2022, https://qldspatial.information.qld.gov.au/catalogue/custom/detail.page?fid={F5CF90D6-5881-4D8F-9581-D8F55D25F9CE}

Department of Environment and Science 2022c, *Remnant vegetation cover by subregion*, State of Queensland, Brisbane, viewed 17/10/2022, <https://www.publications.qld.gov.au/dataset/7e361326-a615-44ac-ba99-a97c75b96fc9/resource/9371bc4d-94b4-4cae-a47e-a52a6d590eef/download/remnant-cover-v12.2-by-subregion.xlsx>.

Department of Natural Resources, Mines and Energy 2019, *Guideline for determining category C areas*, State of Queensland, Brisbane.

Donaghy, P, Bray, S, Gowen, R, Rolfe, J, Stephens, M, Hoffmann, M & Stunzer, A 2010, 'The bioeconomic potential for agroforestry in Australia’s northern grazing systems', *Small-scale Forestry*, vol. 9, no. 4, pp. 463-84.

Dwyer, JM, Fensham, RJ, Butler, DW & Buckley, YM 2009, 'Carbon for conservation: Assessing the potential for win-win investment in an extensive Australian regrowth ecosystem', *Agriculture, Ecosystems and Environment*, vol. 134, no. 1-2, pp. 1-7.

Edwards-Callaway, LN, Cramer, MC, Cadaret, CN, Bigler, EJ, Engle, TE, Wagner, JJ & Clark, DL 2021, 'Impacts of shade on cattle well-being in the beef supply chain', *Journal of animal science*, vol. 99, no. 2, p. skaa375.

Environment Protection and Biodiversity Conservation Act 1999 (Cwth)

Eyre, TJ, Maron, M, Mathieson, MT & Haseler, M 2009, 'Impacts of grazing, selective logging and hyper-aggressors on diurnal bird fauna in intact forest landscapes of the Brigalow Belt, Queensland', *Austral ecology*, vol. 34, no. 6, pp. 705-16.

Ford, HA, Barrett, GW, Saunders, DA & Recher, HF 2001, 'Why have birds in the woodlands of Southern Australia declined?', *Biological Conservation*, vol. 97, no. 1, pp. 71-88.

Gagic, V, Paull, C & Schellhorn, NA 2018, 'Ecosystem service of biological pest control in Australia: the role of non-crop habitats within landscapes', *Austral Entomology*, vol. 57, no. 2, pp. 194-206.

Genstat 2022, *Genstat for Windows, Release 22.1.*, VSN International Ltd, Oxford.

Gibson, E 2022, *Agriculture Biodiversity Stewardship Market Bill 2022*, Department of Parlimentary Services, Parliament of Australia, <https://parlinfo.aph.gov.au/parlInfo/download/legislation/billsdgs/8494211/upload_binary/8494211.pdf>.

Gowen, R & Bray, SG 2016, 'Bioeconomic modelling of woody regrowth carbon offset options in productive grazing systems', *The Rangeland Journal*, vol. 38, no. 3, pp. 307-17.

Graham, L, Gaulton, R, Gerard, F & Staley, JT 2018, 'The influence of hedgerow structural condition on wildlife habitat provision in farmed landscapes', *Biological Conservation*, vol. 220, pp. 122-31.

Green, RE, Osborne, PE & Sears, EJ 1994, 'The Distribution of Passerine Birds in Hedgerows During the Breeding Season in Relation to Characteristics of the Hedgerow and Adjacent Farmland', *The Journal of applied ecology*, vol. 31, no. 4, pp. 677-92.

Grosenbaugh, LR 1952, *Shortcuts for cruisers and scalers*, USA Forest Service, Southern Forest Experiment Station.

Guiller, G, Legentilhomme, J, Boissinot, A, Blouin-Demers, G, Barbraud, C & Lourdais, O 2022 'Response of farmland reptiles to agricultural intensification: Collapse of the common adder Vipera berus and the western green lizard Lacerta bilineata in a hedgerow landscape', *Animal conservation*, vol. 25, no. 5.

Haas, CA 1995, 'Dispersal and use of corridors by birds in wooded patches on an agricultural landscape', *Conservation Biology*, vol. 9, no. 4, pp. 845-54.

Hall, M, Nimmo, D, Watson, S & Bennett, AF 2018, 'Linear habitats in rural landscapes have complementary roles in bird conservation', *Biodiversity and conservation*, vol. 27, no. 10, pp. 2605-23.

Haslem, A, Clarke, RH, Holland, GJ, Radford, JQ, Stewart, A & Bennett, AF 2021, 'Local management or wider context: What determines the value of farm revegetation plantings for birds?', *The Journal of applied ecology*, vol. 58, no. 11, pp. 2552-65.

Hastings, RA & Beattie, AJ 2006, 'Stop the bullying in the corridors: Can including shrubs make your revegetation more Noisy Miner free?', *Ecological Management & Restoration*, vol. 7, no. 2, pp. 105-12.

Heinrich, VHA, Dalagnol, R, Cassol, HLG, Rosan, TM, de Almeida, CT, Silva Junior, CHL, Campanharo, WA, House, JI, Sitch, S, Hales, TC, Adami, M, Anderson, LO & Aragão, LEOC 2021, 'Large carbon sink potential of secondary forests in the Brazilian Amazon to mitigate climate change', *Nature communications*, vol. 12, no. 1, p. 1785.

Hinsley, SA & Bellamy, PE 2000, 'The influence of hedge structure, management and landscape context on the value of hedgerows to birds: A review', *Journal of Environmental Management*, vol. 60, no. 1, pp. 33-49.

Howes, A, Mac Nally, R, Loyn, R, Kath, J, Bowen, M, McAlpine, C & Maron, M 2014, 'Foraging guild perturbations and ecological homogenization driven by a despotic native bird species', *Ibis*, vol. 156, no. 2, pp. 341-54.

Howes, AL & Maron, M 2009, 'Interspecific competition and conservation management of continuous subtropical woodlands', *Wildlife Research*, vol. 36, no. 7, pp. 617-26.

Iwasaki, K, Torita, H, Touze, M, Wada, H & Abe, T 2021, 'Modeling optimal windbreak design in maize fields in cool humid climates: Balancing between positive and negative effects on yield', *Agricultural and Forest Meteorology*, vol. 308-309, p. 108552.

Jamieson, LE, Dobson, S, Cave, J & Stevens, PS 2002, 'A survey of armoured scale insects on kiwifruit shelter', *New Zealand Plant Protection*, vol. 55, no. 0, pp. 354-60.

Jellinek, S, Parris, KM, McCarthy, MA, Wintle, BA & Driscoll, DA 2014, 'Reptiles in restored agricultural landscapes: the value of linear strips, patches and habitat condition: Reptiles in restored agricultural landscapes', *Animal conservation*, vol. 17, no. 6, pp. 544-54.

Johnson, RW 1964, *Ecology and control of brigalow in Queensland*, Queensland Department of Primary Industries, Brisbane.

Kearney, SG, Carwardine, J, Reside, AE, Adams, VM, Nelson, R, Coggan, A, Spindler, R & Watson, JEM 2022, 'Saving species beyond the protected area fence: Threats must be managed across multiple land tenure types to secure Australia's endangered species', *Conservation Science and Practice*.

Lindenmayer, DB, Blanchard, W, Crane, M, Michael, D & Florance, D 2018, 'Size or quality. What matters in vegetation restoration for bird biodiversity in endangered temperate woodlands?', *Austral ecology*, vol. 43, no. 7, pp. 798-806.

Lindenmayer, DB, Knight, EJ, Crane, MJ, Montague-Drake, R, Michael, DR & MacGregor, CI 2010, 'What makes an effective restoration planting for woodland birds?', *Biological Conservation*, vol. 143, no. 2, pp. 289-301.

Liu, Y, Li, H, Yuan, F, Shen, L, Wu, M, Li, W, Wang, A, Wu, J & Guan, D 2022, 'Estimating the impact of shelterbelt structure on corn yield at a large scale using Google Earth and Sentinel 2 data', *Environmental Research Letters*, vol. 17, no. 4, p. 044060.

Lucas, RM, Clewley, D, Accad, A, Butler, D, Armston, J, Bowen, M, Bunting, P, Carreiras, J, Dwyer, J, Eyre, T, Kelly, A, McAlpine, C, Pollock, S & Seabrook, L 2014, 'Mapping forest growth and degradation stage in the Brigalow Belt Bioregion of Australia through integration of ALOS PALSAR and Landsat-derived foliage projective cover data', *Remote sensing of environment*, vol. 155, pp. 42-57.

Mac Nally, R, Bowen, M, Howes, A, McAlpine, CA & Maron, M 2012, 'Despotic, high-impact species and the subcontinental scale control of avian assemblage structure', *Ecology*, vol. 93, no. 3, pp. 668-78.

MacDonald, MA & Kirkpatrick, JB 2003, 'Explaining bird species composition and richness in eucalypt-dominated remnants in subhumid Tasmania', *Journal of biogeography*, vol. 30, no. 9, pp. 1415-26.

Major, RE, Christie, FJ & Gowing, G 2001, 'Influence of remnant and landscape attributes on Australian woodland bird communities', *Biological Conservation*, vol. 102, no. 1, pp. 47-66.

Mappin, B, Ward, A, Hughes, L, Watson, JEM, Cosier, P & Possingham, HP 2022, 'The costs and benefits of restoring a continent's terrestrial ecosystems', *Journal of Applied Ecology*, vol. 59, no. 2, pp. 408-19.

Maron, M 2009, 'Nesting, foraging and aggression of Noisy Miners relative to road edges in an extensive Queensland forest', *Emu*, vol. 109, no. 1, pp. 75-81.

Maron, M, Bowen, M, Fuller, RA, Smith, GC, Eyre, TJ, Mathieson, M, Watson, JEM & McAlpine, CA 2012, 'Spurious thresholds in the relationship between species richness and vegetation cover: Spurious thresholds in species-area relationships', *Global ecology and biogeography*, vol. 21, no. 6, pp. 682-92.

Maron, M, Goulding, W, Ellis, RD & Mohd-Taib, F-S 2012, 'Distribution and individual condition reveal a hierarchy of habitat suitability for an area-sensitive passerine', *Biodiversity and conservation*, vol. 21, no. 10, pp. 2509-23.

Maron, M, Grey, MJ, Catterall, CP, Major, RE, Oliver, DL, Clarke, MF, Loyn, RH, Mac Nally, R, Davidson, I & Thomson, JR 2013, 'Avifaunal disarray due to a single despotic species', *Diversity and Distributions*, vol. 19, no. 12, pp. 1468-79.

Maron, M & Kennedy, S 2007, 'Roads, fire and aggressive competitors: Determinants of bird distribution in subtropical production forests', *Forest ecology and management*, vol. 240, no. 1, pp. 24-31.

Maron, M & Lill, A 2005, 'The influence of livestock grazing and weed invasion on habitat use by birds in grassy woodland remnants', *Biological Conservation*, vol. 124, no. 4, pp. 439-50.

Martin, TG & McIntyre, S 2007, 'Impacts of livestock grazing and tree clearing on birds of woodland and riparian habitats', *Conservation Biology*, vol. 21, no. 2, pp. 504-14.

Martin, TG & Possingham, H 2005, 'Predicting the impact of livestock grazing on birds using foraging height data', *Journal of Applied Ecology*, vol. 42, no. 2, pp. 400-8.

Mayrinck, RC, Laroque, CP, Amichev, BY & Van Rees, K 2019, 'Above- and Below-Ground Carbon Sequestration in Shelterbelt Trees in Canada: A Review', *Forests*, vol. 10, no. 10, p. 922.

McAlpine, CA, Grigg, GC, Mott, JJ & Sharma, P 1999, 'Influence of Landscape Structure on Kangaroo Abundance in a Disturbed Semi-Arid Woodland of Queensland', *The Rangeland Journal*, vol. 21, no. 1, pp. 104-34.

McKeon, G, Chilcott, C, McGrath, W, Paton, C, Fraser, G, Stone, G & Ryan, J 2008, *Assessing the value of trees in sustainable grazing systems*, Meat and Livestock Australia, North Sydney.

Meinke, H, Carberry, PS, Cleugh, HA, Poulton, PL & Hargreaves, JNG 2002, 'Modelling crop growth and yield under the environmental changes induced by windbreaks 1. Model development and validation', *Australian journal of experimental agriculture*, vol. 42, no. 6, pp. 875-85.

Melton, CB, Reside, AE, Simmonds, JS, McDonald, PG, Major, RE, Crates, R, Catterall, CP, Clarke, MF, Grey, MJ, Davitt, G, Ingwersen, D, Robinson, D & Maron, M 2021, 'Evaluating the evidence of culling a native species for conservation benefits', *Conservation Science and Practice*, vol. 3, no. 12, p. n/a.

Meat and Livestock Australia 2020, *The Australian Red Meat Industry's Carbon Neutral by 2030,* Sydney.

Montague-Drake, RM, Lindenmayer, DB & Cunningham, RB 2009, 'Factors affecting site occupancy by woodland bird species of conservation concern', *Biological Conservation*, vol. 142, no. 12, pp. 2896-903.

Mukul, SA, Herbohn, J & Firn, J 2016, 'Co-benefits of biodiversity and carbon sequestration from regenerating secondary forests in the Philippine uplands: implications for forest landscape restoration', *Biotropica*, vol. 48, no. 6, pp. 882-9.

Murcia, C 1995, 'Edge effects in fragmented forests: implications for conservation', *Trends in Ecology & Evolution*, vol. 10, no. 2, pp. 58-62.

Neldner, VJ, Niehus, RE, Wilson, BA, McDonald, WJF, Ford, AJ & Accad, A 2019, *The Vegetation of Queensland. Descriptions of Broad Vegetation Groups.*, Queensland Herbarium, State of Queensland.

Osorio, RJ, Barden, CJ & Ciampitti, IA 2019, 'GIS approach to estimate windbreak crop yield effects in Kansas–Nebraska', *Agroforestry systems*, vol. 93, no. 4, pp. 1567-76.

Pannell, DJ, Marshall, GR, Barr, N, Curtis, A, Vanclay, F & Wilkinson, R 2006, 'Understanding and promoting adoption of conservation practices by rural landholders', *Australian journal of experimental agriculture*, vol. 46, no. 11, pp. 1407-24.

Piper, SD & Catterall, CP 2003, 'A particular case and a general pattern: hyperaggressive behaviour by one species may mediate avifaunal decreases in fragmented Australian forests', *Oikos*, vol. 101, no. 3, pp. 602-14.

Piper, SD & Catterall, CP 2004, 'Effects of edge type and nest height on predation of artificial nests within subtropical Australian eucalypt forests', *Forest ecology and management*, vol. 203, no. 1, pp. 361-72.

Ponce Reyes, R, Firn, J, Nicol, S, Chadès, I, Stratford, D, Martin, T, Whitten, S & Carwardine, J 2016, *Priority Threat Management for Imperilled Species of the Queensland Brigalow Belt*, CSIRO, Brisbane.

Queensland Globe 2022, *Broad vegetation groups*, State of Queensland, viewed at 28/10/2022 <https://qldglobe.information.qld.gov.au/>.

Radford, JQ, Bennett, AF & Cheers, GJ 2005, 'Landscape-level thresholds of habitat cover for woodland-dependent birds', *Biological Conservation*, vol. 124, no. 3, pp. 317-37.

Rempel, JC, Kulshreshtha, SN, Amichev, BY & Rees, KCJV 2017, 'Costs and benefits of shelterbelts: A review of producers' perceptions and mind map analyses for Saskatchewan, Canada', *Canadian Journal of Soil Science*, vol. 97, no. 3, pp. 341-52.

Robertson, OJ & Radford, jQ 2009, 'Gap-crossing decisions of forest birds in a fragmented landscape', *Austral ecology*, vol. 34, no. 4, pp. 435-46.

RStudio_Team 2022, *RStudio: Integrated Development Environment for R*, PBC, Boston, <http://www.rstudio.com/>.

Ryan, JG, McAlpine, CA & Ludwig, JA 2010, 'Integrated vegetation designs for enhancing water retention and recycling in agroecosystems', *Landscape Ecology*, vol. 25, no. 8, pp. 1277-88.

Šálek, M, Bažant, M, Żmihorski, M & Gamero, A 2022, 'Evaluating conservation tools in intensively-used farmland: Higher bird and mammal diversity in seed-rich strips during winter', *Agriculture, ecosystems & environment*, vol. 327, p. 107844.

Saunders, DA & de Rebeira, CP 1991, 'Values of corridors to avian populations in a fragmented landscape', in DA Saunders & RJ Hobbs (eds), *Nature Conservation 2: The role of corridors*, Surrey Beatty & Sons, Chipping Norton, NSW, pp. 221-40.

Saunders, DA & Hobbs, RJ 1991, *The role of corridors*, Surrey Beatty, Chipping Norton, N.S.W.

Scanlan, JC 1991, 'Woody overstorey and herbaceous understorey biomass in Acacia harpophylla (brigalow) woodlands Queensland', *Australian Journal of Ecology*, vol. 16, no. 4, pp. 521-9.

Scanlan, JC 2002, 'Some aspects of tree-grass dynamics in Queensland's grazing lands', *The Rangeland Journal*, vol. 24, no. 1, pp. 56-82.

Scanlan, JC & Anderson, ER 1981, 'Use of the heavy duty blade plough for control of woody regrowth in Central Queensland ', *Australian Weeds*, pp. 10-2.

Scherr, SJ & McNeely, JA 2008, 'Biodiversity conservation and agricultural sustainability: towards a new paradigm of ‘ecoagriculture’ landscapes', *Philosophical Transactions of the Royal Society B: Biological Sciences*, vol. 363, no. 1491, pp. 477-94.

Seabrook, L, McAlpine, C & Fensham, R 2006, 'Cattle, crops and clearing: Regional drivers of landscape change in the Brigalow Belt, Queensland, Australia, 1840–2004', *Landscape and Urban Planning*, vol. 78, no. 4, pp. 373-85.

Selwood, K, Mac Nally, R & Thomson, JR 2009, 'Native bird breeding in a chronosequence of revegetated sites', *Oecologia*, vol. 159, no. 2, pp. 435-46.

Simmonds, JS, Rensburg, BJ, Tulloch, AIT, Maron, M & Willis, S 2019, 'Landscape-specific thresholds in the relationship between species richness and natural land cover', *The Journal of applied ecology*, vol. 56, no. 4, pp. 1019-29.

Sklenicka, P, Molnarova, K, Brabec, E, Kumble, P, Pittnerova, B, Pixova, K & Salek, M 2009, 'Remnants of medieval field patterns in the Czech Republic: Analysis of driving forces behind their disappearance with special attention to the role of hedgerows', *Agriculture, ecosystems & environment*, vol. 129, no. 4, pp. 465-73.

Smith, IM, Hoffmann, AA & Thomson, LJ 2015, 'Ground cover and floral resources in shelterbelts increase the abundance of beneficial hymenopteran families', *Agricultural and Forest Entomology*, vol. 17, no. 2, pp. 120-8.

Sullivan, MJP, Pearce-Higgins, JW, Newson, SE, Scholefield, P, Brereton, T, Oliver, TH & McKenzie, A 2017, 'A national-scale model of linear features improves predictions of farmland biodiversity', *J Appl Ecol*, vol. 54, no. 6, pp. 1776-84.

Swinton, SM, Lupi, F, Robertson, GP & Hamilton, SK 2007, 'Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits', *Ecological Economics*, vol. 64, no. 2, pp. 245-52.

Thomson, JR, Maron, M, Grey, MJ, Catterall, CP, Major, RE, Oliver, DL, Clarke, MF, Loyn, RH, Davidson, I, Ingwersen, D, Robinson, D, Kutt, A, MacDonald, MA & Mac Nally, R 2015, 'Avifaunal disarray: quantifying models of the occurrence and ecological effects of a despotic bird species', *Diversity Distrib*, vol. 21, no. 4, pp. 451-64.

Tunstall, BR & Connor, DJ 1981, 'hydrological study of a subtropical semiarid forest of Acacia harpophylla F. Muell. ex Benth. (Brigalow)', *Australian Journal of Botany*, vol. 29, no. 3, pp. 311-20.

Usieta, HO, Manu, SA & Ottosson, U 2013, 'Farmland conservation in West Africa: how do hedgerow characteristics affect bird species richness?', *Bird study*, vol. 60, no. 1, pp. 102-10.

van der Ree, R & Bennett, AF 2003, 'Home range of the squirrel glider (Petaurus norfolcensis) in a network of remnant linear habitats', *J. Zoology*, vol. 259, no. 4, pp. 327-36.

Vesk, PA, Nolan, R, Thomson, JR, Dorrough, JW & Nally, RM 2008, 'Time lags in provision of habitat resources through revegetation', *Biological Conservation*, vol. 141, no. 1, pp. 174-86.

Woinarski, JCZ, McCosker, JC, Gordon, G, Lawrie, B, James, C, Augusteyn, J, Slater, L & Danvers, T 2006, 'Monitoring change in the vertebrate fauna of central Queensland, Australia, over a period of broad-scale vegetation clearance, 19732002', *Wildlife Research*, vol. 33, no. 4, pp. 263-74.

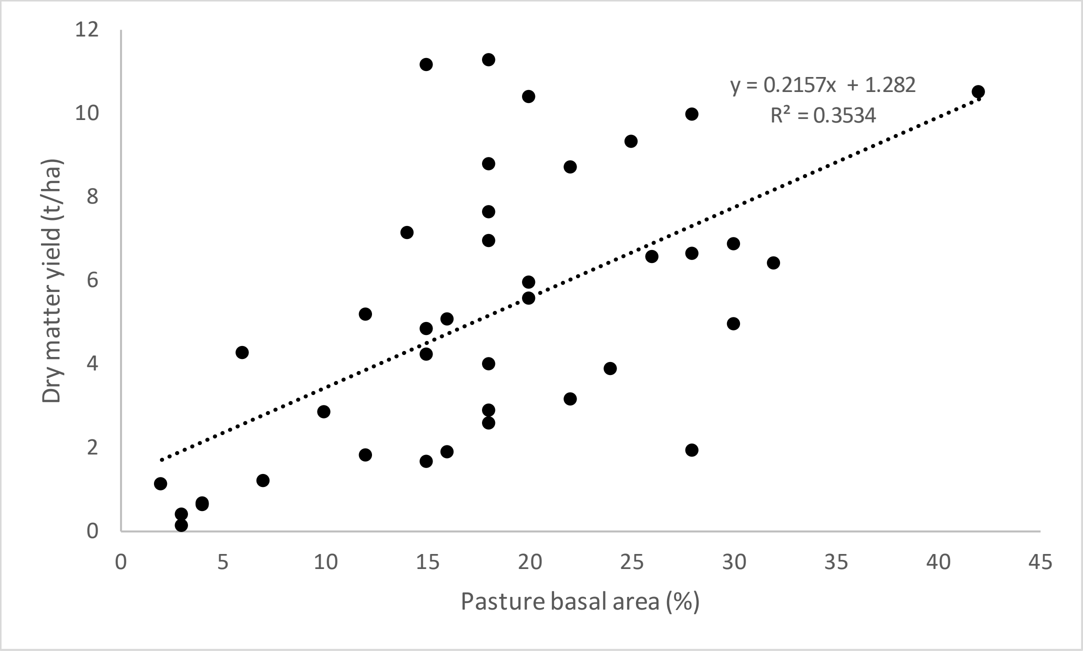
Young, KJ 2017, 'Mimicking Nature: A Review of Successional Agroforestry Systems as an Analogue to Natural Regeneration of Secondary Forest Stands', in F Montagnini (ed.), *Integrating Landscapes: Agroforestry for Biodiversity Conservation and Food Sovereignty*, Springer International Publishing, Cham, pp. 179-209.

Young, L & Schlesinger, C 2015, 'Habitat use and behaviour of birds in areas invaded by buffel grass Cenchrus ciliaris and in restored habitat', *Wildlife Research*, vol. 41, no. 5, pp. 379-94.

Zurita, G, Pe'er, G, Bellocq, MI & Hansbauer, MM 2012, 'Edge effects and their influence on habitat suitability calculations: a continuous approach applied to birds of the Atlantic forest', *The Journal of applied ecology*, vol. 49, no. 2, pp. 503-12.

SUPPLEMENTARY INFORMATION

a)



b)

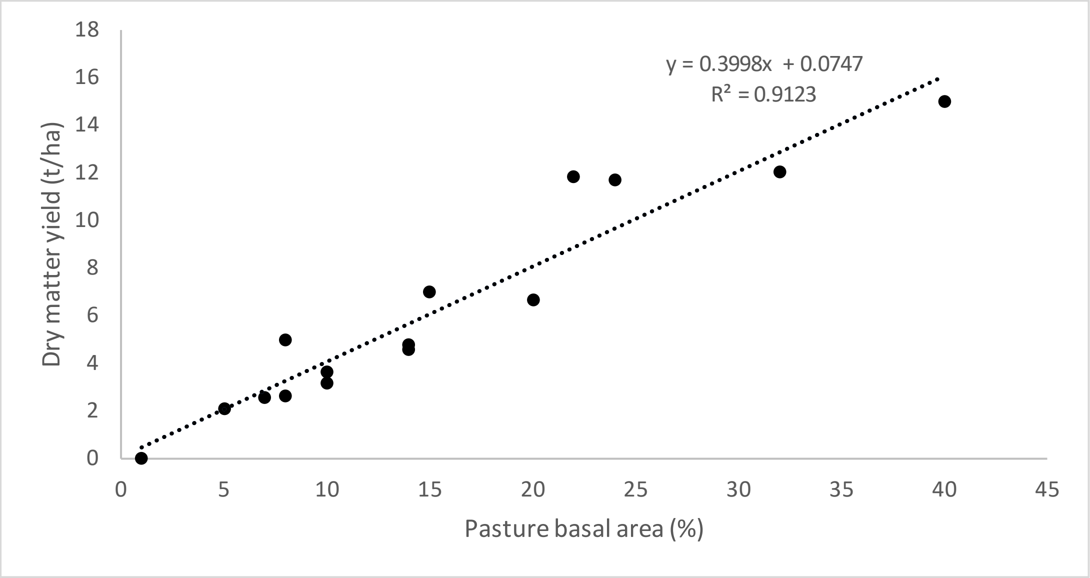


Figure S1 The relationship between pasture basal area and dry matter yield for **a)** *Cenchrus ciliaris* var. *Gayndah* and **b)** *Cenchrus ciliaris* var. *Biloela.* Data points represent individual 0.5 x 0.5 m quadrats.

Table S1 List of birds detected during this study (n= 54). 32 were classified as woodland dependant. Foraging guilds included granivores (G), nectarivores/frugivores (NF), ground insectivores (Ig), canopy insectivores (Ic), mixed insectivores (Im), carnivores (C). Woodland bird and foraging guild classifications were based off Bowen et al. (2009).

Common Name	Scientific Name	Foraging guild	Woodland dependent	Total sites
Singing Honeyeater	*Lichenostomus virescens*	NF	WD	29
Grey Butcherbird	*Cracticus torquatus*	Im	WD	25
Yellow Thornbill	*Acanthiza nana*	Ic	WD	23
Rufous Whistler	*Pachycephala rufiventris*	Ic	WD	23
Willie Wagtail	*Rhipidura leucophrys*	Ig		22
Inland thornbill	*Acanthiza apicalis*	Ic	WD	21
Striped Honeyeater	*Plectorhyncha lanceolata*	NF	WD	21
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	Ig		21
Spiny-cheeked honeyeater	*Acanthagenys rufogularis*	NF	WD	18
Grey Fantail	*Rhipidura fuliginosa*	Ic	WD	18
Noisy Miner	*Manorina melanocephala*	NF		16
Weebill	*Smicrornis brevirostris*	Ic	WD	12
Pied Butcherbird	*Cracticus nigrogularis*	Im		12
Pale-headed Rosella	*Platycercus adscitus*	G		12
Australian Magpie	*Gymnorhina tibicen*	Ig		11
Double-barred Finch	*Taeniopygia bichenovii*	G		9
Torresian Crow	*Corvus orru*	O		8
Superb Fairy Wren	*Malurus cyaneus*	Ig		8
Chestnut-rumped thornbill	*Acanthiza uropygialis*	Ic	WD	7
Speckled Warbler	*Chthonicola sagittate*	Ig	WD	7
Mistletoe bird	*Dicaeum hirundinaceum*	NF	WD	7
Noisy Friarbird	*Philemon corniculatus*	NF	WD	6
Little Friarbird	*Philemon citreogularis*	NF	WD	5
Grey Shrike-Thrush	*Colluricincla harmonica*	Im	WD	5
Crested Pigeons	*Ocyphaps lophotes*	G		5
Red-winged Parrot	*Aprosmictus erythropterus*	G		4
Varied Sitella	*Daphoenositta chrysoptera*	Ic	WD	3

Grey-crowned babbler	*Pomatostomus temporalis*	Im	WD	3
Eastern Yellow Robin	*Eopsaltria australis*	Im	WD	3
Leaden flycatcher	*Myiagra rubecula*	Ic	WD	3
Variegated Fairy Wren	*Malurus lamberti*	Ig		3
Apostlebird	*Struthidea cinerea*	Ig		3
silver_eye	*Zosterops lateralis*	NF	WD	2
Brown Goshawk	*Accipiter fasciatus*	C	WD	2
White-browed babbler	*Pomatostomus superciliosus*	Im	WD	2
Common Bronzewing	*Phaps chalcoptera*	G	WD	2
Black-eared Cuckoo	*Chrysococcyx osculans*	Ic	WD	2
Red-capped Robin	*Petroica goodenovii*	Ig	WD	2
Laughing Kookaburra	*Dacelo novaeguineae*	Ic	WD	2
Yellow-throated miner	*Manorina flavigula*	NF		2
Black-shouldered Kite	*Elanus axillaris*	C		2
Black-faced Cuckoo shrike	*Coracina novaehollandiae*	Ic		2
Brown Quail	*Synoicus ypsilophorus*	G		2
Brown-headed honeyeater	*Melithreptus brevirostris*	NF	WD	1
White Plumed Honeyeater	*Lichenostomus penicillatus*	NF	WD	1
Blue-faced Honeyeater	*Entomyzon cyanotis*	NF	WD	1
Yellow-faced honyeater	*Lichenostomus chrysops*	NF	WD	1
Bar-shouldered Dove	*Geopelia humeralis*	G	WD	1
Striated Pardalote	*Pardalotus striatus*	Ic	WD	1
Magpie-lark	*Grallina cyanoleuca*	Ig		1
Blue Bonnet	*Northiella haematogaster*	G		1
Galah	*Eolophus roseicapilla*	G		1
Brown Falcon	*Falco berigora*	C		1
Eastern Barn Owl	*Tyto alba*	C		1

Table S2 Results of the model selection analyses for woodland bird richness and abundance, and *Manorina* spp. abundance

Model		Model variables	Log(L)	AICc	Δ_i	ω_i
Woodland bird richness						
	1	Buffel, *Manorina*, width	-106.15	226.4	0	0.19
	2	*Manorina,* veg status, width	-106.39	226.88	0.47	0.15
	3	Buffel, Manorina, veg status, width	-105.25	227.37	0.96	0.12
	4	Buffel, *Manorina*, veg status	-107.03	228.15	1.75	0.08
	5	*Manorina,* width	-108.36	228.19	1.78	0.08
	6	Buffel, *Manorina,* Rem. and HVR cover, veg status	-105.67	228.21	1.81	0.08
	7	Buffel, Manorina, Rem. and HVR cover, width	-106.13	229.13	2.72	0.05
	8	*Manorina*, Rem. and HVR cover, veg status	-107.61	229.32	2.91	0.05
	9	*Manorina*, Rem. and HVR cover, veg status, width	-106.3	229.47	3.07	0.04
	10	Buffel*, Manorina*	-109.12	229.7	3.29	0.04
	11	Buffel, *Manorina,* Rem. and HVR cover, veg status, width	-105.03	229.84	3.44	0.03
	12	*Manorina*, veg status	-109.3	230.06	3.65	0.03
	13	*Manorina,* Rem. and HVR cover, width	-108	230.11	3.7	0.03
	14	Buffel*, Manorina,* Rem. and HVR cover	-108.09	230.28	3.88	0.03
Woodland bird abundance						
	1	*Manorina,* width	-178.34	368.14	0	0.37
	2	*Manorina,* Rem. and HVR cover, width	-177.8	369.71	1.57	0.17
	3	Buffel, *Manorina,* width	-177.88	369.86	1.72	0.16
	4	*Manorina,* veg status, width	-178.36	370.83	2.69	0.1
	5	Width	-181.07	371.09	2.95	0.08
	6	Buffel, *Manorina,* Rem. and HVR cover, width	-177.37	371.61	3.47	0.07
	7	*Manorina,* Rem. and HVR cover, veg status, width	-177.47	371.8	3.66	0.06
Manorina spp. abundance						
	1	Veg density	-93.71	196.37	0	0.19
	2	Veg density, coarse debris	-92.73	196.92	0.55	0.14
	3	Veg density, width	-92.77	197.01	0.63	0.14
	4	Veg density, width, coarse debris	-91.76	197.62	1.25	0.1
	5	Pasture height, veg density	-93.48	198.43	2.06	0.07
	6	Rem. and HVR cover, veg density	-93.69	198.83	2.46	0.06
	7	Rem. and HVR cover, veg density, width	-92.47	199.05	2.68	0.05
	8	Pasture height, veg density, coarse debris	-92.6	199.31	2.94	0.04
	9	Pasture height, veg density, width	-92.63	199.36	2.99	0.04
	10	Width	-95.24	199.43	3.06	0.04
	11	Rem. and HVR cover, veg density, coarse debris	-92.72	199.53	3.16	0.04
	12	Rem. and HVR cover, veg density, width, coarse debris	-91.47	199.8	3.43	0.03
	13	Rem. and HVR cover, width	-94.31	200.08	3.71	0.03
	14	Pasture height, veg density, width, coarse debris	-91.7	200.27	3.9	0.03

Values represent the maximized log-likelihood (Log(L)), Akaike information criteria corrected for small sample size (AIC$_c$), the changes in AIC$_c$ (Δ_i) and the Akaike weight of each model (ω_i). Table design modified from Eyre et al. 2009, pg. 712.

Table S3 Akaike weights (w_i) calculated from AICc of alternative models of strip width as a function of each of the six explanatory variables. Bolded models are significant ($P<0.05$), and those denoted with an * were the best performing models.

Model	Vegetation structural complexity		Buffel		Mistletoe		Large trees		Vegetation density		Coarse woody debris	
	AICc	w_i	AICc	w_i	AICc	w_i	AICc	w_i	AICc	w_i	AICc	w_i
Linear	**36.27***	**0.39***	268.73	0.37	531.03	0.39	**328.97**	**0.46**	904.85	0.38*	665.04	0.56*
Exponential	**37.02**	**0.27**	267.68	0.63*	530.1	0.61*	**328.68***	**0.53***	905.22	0.32	665.50	0.44
Power	**36.52**	**0.34**	n.a.	n.a.	n.a.	n.a.	n.a.	n.a.	905.34	0.30	n.a.	n.a.