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3	Biodiversity benefits of vegetation restoration are undermined by livestock grazing
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18	
19	Running Head: Grazing effects on biodiversity in plantings
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21 Abstract

22 Extensive areas of the Earth's terrestrial surface have been subject to restoration, but how best to manage such restored areas has received relatively limited attention. Here we 23 24 quantify the effects of livestock grazing on bird and reptile biota within 61 restoration plantings in south-eastern Australia. Using path analysis, we identified some of the 25 mechanisms giving rise to differences in patterns of species richness and individual species 26 occurrence between grazed and ungrazed plantings. Specifically, we found evidence of both: 27 (1) indirect effects of grazing on various elements of biodiversity mediated through changes 28 29 in vegetation condition (primarily the leaf litter layer), and (2) direct effects of grazing on biodiversity (irrespective of modification in vegetation cover attributes), possibly as a result 30 of trampling by livestock. We also uncovered evidence of direct effects on bird and reptile 31 32 biota of other planting attributes such as planting width and planting age. The results of our study suggest the biodiversity benefits of restoration programs can be undermined by grazing, 33 especially by uncontrolled grazing. We suggest that where the objective of vegetation 34 35 restoration is to enhance biodiversity conservation, grazing within plantings should be limited or excluded. 36

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Keywords: Livestock grazing, restoration planting, south-eastern Australia, woodland birds,
woodland reptiles, woodland restoration

40 Implications for Practice

Livestock grazing in restored (replanted) areas should be limited or excluded to minimize
negative impacts on birds and reptiles.

In particular, uncontrolled grazing should be avoided wherever possible as it can have
direct impacts on biodiversity (e.g. via trampling) and/or indirect effects mediated through
altered ground layer conditions.

A key implication for management is that fences should be maintained so that access to
replanted vegetation by livestock can be either be excluded or the amount of grazing
pressure can be controlled.

In addition to grazing control, other key attributes of plantings that can have significant
impacts on the effectiveness for biodiversity and on which managers can have a direct
influence include planting width and the age of plantings.

52 Introduction

Billions of dollars are being spent annually by goverments and organizations to 53 restore the millions of hectares of degraded land worldwide (Hajkowicz 2009; Aronson & 54 Alexander 2013; Menz et al. 2013; Kimball et al. 2015; Crouzeilles et al. 2016). A wide 55 range studies as well as extensive meta-analyses have highlighted the biodiversity benefits of 56 restoration efforts (e.g. Gibb & Cunningham 2010; Crouzeilles et al. 2016; Lindenmayer et 57 58 al. 2016). However, how restored areas are managed may influence their biodiversity values. For example, livestock grazing may influence the habitat value of restored vegetation, 59 60 especially as it is the largest user of agricultural land globally (FAO 2009). Livetsock grazing can alter the amount and structure of vegetation cover (Lunt et al. 2007; Sato et al. 2016) and 61 influence the suitability of habitats for biodiversity (Williams & Price 2011; Lindenmayer et 62 al. 2012). Grazing also can affect the composition of the vegetation, although this was not a 63 64 focus of the article reported here. Whilst the effects of grazing on the biota inhabiting remnant vegetation is relatively well understood (e.g. Kay et al. 2017), its effects on biota in 65 restored areas have only rarely been examined. 66

67 Here we report the findings of an empirical study of grazing in restored areas in the 68 South West Slopes of New South Wales, south-eastern Australia. This region is one of the 69 most heavily modified by agricultural activities in Australia (Benson 2008) and it has been 70 targeted for extensive revegetation efforts over the past 20 years (Crane et al. 2014; 71 Lindenmayer et al. 2016). Indeed, past studies in the South West Slopes have highlighted the 72 value of planted areas for biodiversity (Barrett et al. 2008; Lindenmayer et al. 2010b; Lindenmayer et al. 2016). However, many revegetated areas are being grazed, in part because 73 74 of the state of disrepair of fences around plantings which means they are increasingly accessible to livestock (M. Crane, personal observation). This presents an important 75 opportunity to determine if the value of restored (planted) woodlands for bird and reptile 76 biota is altered by grazing by domestic livestock. The key question which motivated our 77 study was therefore: Is there a difference in bird and reptile biodiversity between grazed and 78 79 ungrazed plantings and, if so, what mechanisms might underpin such grazing impacts? As different groups of biota have different habitat requirements, our investigation explored 80 relationships between both birds reptiles and vegetation structure. 81

82 We underpinned our work with a conceptual model of the potential direct and indirect inter-relationships between planting attributes (e.g. width and age), grazing, vegetation 83 condition, and the species richness and occurrence of birds and reptiles (Fig. 1). Previous 84 85 studies have highlighted the impacts of grazing on the understory and other layers of vegetation (Spooner et al. 2002; Lunt et al. 2007; Martin & McIntyre 2007; Lindenmayer et 86 al. 2012; Sato et al. 2016), albeit in remnant woodland patches and not in restored areas. 87 Other work has produced evidence of the effects of livestock grazing on groups such as birds 88 (Martin & McIntyre 2007; Lindenmayer et al. 2012) and reptiles (e.g. Kay et al. 2017; 89 90 Pulsford et al. 2017), again in eucalypt remnants rather than plantings. We sought to quantify both indirect and direct effects of grazing on the biota inhabiting plantings. For example, 91 indirect effects of livestock grazing on biodiversity may be mediated by grazing-related 92 93 modification of vegetation cover (Fleishman & Murphy 2009; Williams & Price 2011) that, in turn, alters habitat suitability for birds and reptiles (Martin & McIntyre 2007). Direct 94 effects of grazing may occur without intermediary impacts on vegetation change such as 95

96 through livestock trampling of bird nests located on the ground (Higgins 1991-2006;
97 Williams & Price 2011). Direct effects of planting attributes may arise because animals

respond to measures like width and/or age irrespective of modification of vegetation
condition that may result from livestock grazing (Fig. 1). These indirect and direct effects of
planting attributes and grazing represent different mechanisms or pathways (sensu Shipley
2009) giving rise to patterns of species richness and individual species occurrence.

Millions of hectares of the Earth's terrestrial surface are planned to be targeted in 102 large-scale restoration programs (Menz et al. 2013; Crouzeilles et al. 2016; McAlpine et al. 103 104 2016), in part to tackle problems associated with land degradation but also to address other environmental problems such as biodiversity loss (Aronson & Alexander 2013; Crouzeilles et 105 al. 2016). How restored areas are managed, including grazing management, may have a 106 107 significant influence on their effectiveness for both conserving biodiversity. The work reported in this paper is therefore relevant to the management of the increasing amount of 108 restored native vegetation globally. 109

110 Methods

111 Study area

Our study region was a 150 x 120 km agricultural area within the South West Slopes bioregion of New South Wales, south-eastern Australia. The South West Slopes was formerly dominated by temperate eucalypt woodland (Lindenmayer et al. 2010a), but has been cleared of an estimated 85% of its original cover (Benson 2008) to facilitate livestock grazing and cereal cropping. As a result, the South West Slopes region has been the target of major planting programs (Cunningham et al. 2014).

118 Plantings attributes

We focused on 61 areas of replanted native vegetation on 25 farms in our study region.
A total of 41 plantings has never been grazed by domestic livestock. For the remaining 20

plantings, grazing was "controlled" in that it occurred infrequently (e.g. occasional "crash" grazing) on 10 sites or "uncontrolled" in that cattle (*Bos taurus*) and sheep (*Ovis aries*) had continuous access to plantings on a further 10 sites. The width of plantings ranged from 10-300 m. Age of planting ranged from 6 to 61 years old (25^{th} percentile = 13 years, median = 18 years, and 75th percentile = 23 years) (Table S1).

Our plantings were characterized by a mix of local endemic and exotic Australian 126 ground cover, understory, and overstory plant species. Most plants were typically spaced 2 m 127 apart, but there was not a standard set of spacing and plant species composition protocols 128 129 applied in revegetation efforts. In spring 2013, we completed a survey of vegetation structure and composition in the plantings with the primary focus of this study being on six attributes. 130 These were the percentage cover in the understory, midstory and overstory, the percentage 131 132 cover of leaf litter, the percentage of tussocks of exotic grass, and the percentage of tussocks of native grass. We measured the six vegetation cover variables in three 20 x 20 m plots at 0 133 m, 100 m and 200 m points along a permanent transect at each site. We defined understory, 134 midstory and overstory based on height; the overstory was vegetation exceeding 10 m in 135 height, midstory was 2-10 m in height, and understory was woody vegetation less than 2 m in 136 height. To obtain a site-level description of the vegetation cover variables, we aggregated 137 plot-level data to the site-level. 138

139 Bird surveys

We gathered bird data in the spring of 2013 using repeated five-minute point interval counts at 0 m, 100 m and 200 m along the fixed transect at each of the 61 plantings. For each point-interval count, an observer recorded all bird species seen or heard within 50 m of the centre of a field plot point. Each site was surveyed twice by two observers on different days. We did not undertake surveys during poor weather (rain, high wind, fog or heavy cloud cover). We observed these protocols to maximize the detection of bird species and reduce the effects of observer heterogeneity and day effects (Lindenmayer et al. 2009). All bird surveys
were completed by the same group of experienced observers from The Australian National
University.

149 *Reptile surveys*

We surveyed reptiles in two ways. First, we completed time-constrained (20 minute) 150 active searches (see Michael et al. 2012). Second, we deployed three kinds of artificial refuge 151 arrays to survey the occurrence of reptile species in each of the 61 plantings. The artificial 152 refuges were: (1) one double-layered stack of corrugated galvanized steel; (2) four 1.2m long 153 154 railway sleepers; and (3) four concrete roof tiles $(32 \times 42 \text{ cm})$ (Michael et al. 2012). At each site, we established two reptile monitoring stations located at the 0 m and 100 m point along 155 the same 200 metre transect on which bird surveys were completed (see above). Surveys 156 157 were conducted during the spring of 2013 and confined to clear sunny days between 09:00 and 14:00 h by experienced herpetologists from The Australian National University. Previous 158 analysis have indicated that the use of an array of search and survey methods ensures that 159 almost all species of reptiles are detected in plantings (see Michael et al. 2012). 160

161 Statistical analysis

Shipley (2008) and the references therein lay out an approach to path analysis based 162 on directed acyclic graphs and the concept of d-separation. We employed Bayesian 163 regression models with paths chosen via leave-one-out cross-validation information criteria 164 (LOOIC) (Gelman et al. 2014). The specific details are described below. Following Shipley 165 (2008), we combined the results of the implied independence claims using Fisher's C 166 statistic, with the frequentist p-values in the definition replaced by their Bayesian 167 168 counterparts. If the data are generated according to the specified causal model then Fisher's C statistic follows a chi-squared distribution with 2c degrees of freedom (where c is the number 169 of implied independence claims; Shipley 2008). Therefore, large values of Fisher's C statistic 170

relative to a chi-squared distribution with 2c degrees of freedom give evidence against thespecified causal model (i.e. small significance levels).

We converted vegetation cover variables (percentage cover of: understory, midstory, 173 overstory, native tussock, exotic tussock and leaf litter) to proportions and modelled them 174 with zero-inflated Beta regression to account for zeros (the Beta distribution is restricted to 175 the open interval (0, 1)). Beta regression was used for midstory cover as no zeros were 176 observed for this variable. Our study design has plantings nested within farms. Therefore, 177 farm was included as a random effect (i.e. we have a multi-level model). The predictor 178 179 variables for this stage of analysis were: age of planting, width of planting (log transformed) and type of grazing (none, controlled and uncontrolled). Note that we examined two 180 additional variants of grazing, grazed vs non-grazed (i.e. we combined controlled and 181 182 uncontrolled grazing) and uncontrolled versus controlled and non-grazed. We used leave one out cross LOOIC (Watanabe 2010; Gelman et al. 2014; Vehtari et al. 2015) for model 183 selection and chose the simplest model (smallest number of terms) within 2 LOOIC units of 184 the best fitting model. 185

We modelled bird and reptile species richness with Poisson regression with a random 186 effect for farm as with vegetation. We modelled the presence/absence of five individual bird 187 species (presence over the 6 point counts) and one species of reptile (any occurrence of the 188 species detected using the various field survey methods) – Rufous Songlark (Megalurus 189 190 mathewsi), Superb Fairy-wren (Malurus cyaneus), Willie Wagtail (Rhipidura leucophrys), White-plumed Honeyeater (Lichenostomus penicillatus), Australian Magpie (Cracticus 191 tibicen), and Boulenger's Skink (Morethia boulengeri) - with logistic regression using farm 192 193 as a random effect. These six taxa were the six most common species with sufficient data to underpin robust path analyses. 194

195 We employed the same site-level characteristics used in the vegetation analysis as predictors in the Poisson and logistic regression models and also used standardized versions 196 of the six vegetation cover variables. Due to the number of predictor variables under 197 198 consideration at this stage, we did not employ an all possible subsets strategy. Instead, we considered the 16 models (Table S2) for the site characteristics combined with the 42 models 199 (Table S3) for the vegetation variables where, at most, three vegetation variables were 200 considered at any one time. We employed LOOIC using a similar strategy as described 201 previously. 202

We used a Bayesian approach, which was implemented using the brms (Bayesian Regression Model Stan) package ((Bürkner 2016) in R (R Core Team 2015)). Continuous predictor variables were standardized prior to entry in the models. We used the brms default priors for the Beta and Poisson modelling, and we employed Cauchy priors (location = 0, scale = 5/2) to control the potential effects of complete separation (see Gelman 2008).

208 **Results**

We recorded 89 species of birds (Table S1) and 15 species of reptiles (Table S2). Of these, no reptile species and two bird species (House Sparrow [*Passer domesticus*] and Common Blackbird [*Turdus merula*]) are exotic taxa.

212 *Covariate effects on vegetation cover*

Descriptive information for all the variables used in the analysis is given in Table S3. In the initial step of our path analysis, we modeled relationships between planting attributes (age, width and type of grazing [*viz*: none, controlled and uncontrolled]) and the various vegetation cover variables. We uncovered evidence of a negative effect of uncontrolled grazing on the amount of leaf litter, a negative effect of grazing compared to no grazing for native tussocks, and a negative effect of planting width (log) on the amount of midstory cover (Tables S4, S6). The null model was the most parsimonious model (within 2 LOOIC units of the best fitting model) for all other vegetation cover variables and combinations of covariates
(Table S4). Notably, there was only limited correlation between the six vegetation cover
variables (the amount of cover in the understory, midstory, overstory, native tussock, exotic
tussock and leaf litter) (see Table S7).

224 Planting attribute and vegetation cover pathway effects on bird and reptile species richness

We found that bird species richness increased with understory cover and the amount 225 of leaf litter (Fig. 2). There also was a negative effect of midstory cover on bird species 226 richness (Tables S8, S9). In addition, there was a direct positive effect of planting width (log) 227 228 on bird species richness (i.e. an effect not mediated through vegetation attributes) (Fig. 2). In the case of reptile species richness, there was evidence of negative effects of uncontrolled 229 grazing and the amount of vegetation cover in the overstory. There also was a positive 230 231 relationship between the amount of leaf litter and reptile species richness (Tables S8, S9; Fig. 3). The causal model for bird species richness fitted the observed data well (Fisher's C =232 36.66, df = 28, tail area = 0.129), however, there some evidence that the reptile species 233 richness model did not fit the observed data well (Fisher's C = 48.01, df = 30, tail area = 234 0.020). The lack of fit of the causal model was caused by a correlation between overstory and 235 midstory (controlling for planting width) (Bayesian tail area =0.001, Table S10). 236

237 Planting attribute and vegetation cover pathway effects on individual bird and reptile

238 species

We completed path analysis for five bird and one reptile species (Tables S8, S9). The Rufous Songlark was negatively associated with grazing (uncontrolled grazing vs no grazing) and positively associated with exotic tussocks (Table S9, Fig. S1). The White-plumed Honeyeater was negatively associated with grazing and overstory cover and positively associated with leaf litter (Table S9, Fig. S2). The Superb Fairy-wren was positively associated with leaf litter (Table S9, Fig. S3). The Australian Magpie and Willie Wagtail

were not associated with any covariates. We found that the Boulenger's Skink was positively 245 associated with leaf litter and the age of planting (see Table S9, Fig. S4). The causal (path) 246 models fit reasonably well for all species with the exception of Rufous Songlark (Fisher's C 247 = 50.97, df = 32, tail area = 0.018) and White-plumed Honeyeater (Fisher's C = 37.37, df = 248 24, tail area = 0.040) (Table S10). The main contribution to the lack of fit of the causal 249 models for these species is correlation between vegetation variables: leaf litter and exotic 250 tussocks controlling for grazing for Rufous Songlark and midstory and overstory cover 251 controlling for planting width for White-plumed Honeyeater. 252

253 Discussion

We found differences in bird and reptile biodiversity between grazed and ungrazed 254 plantings. Moreover, using path analysis, we identified both: (1) indirect effects of grazing on 255 256 various elements of biodiversity as mediated by changes in vegetation condition and (2) direct effects of grazing (irrespective of modification in vegetation attributes). Thus, we were 257 able to identify some of the mechanisms by which grazing can influence the occurrence of 258 bird and reptile biota in plantings. We also uncovered evidence of direct effects on bird and 259 reptile biota of other planting attributes such as planting width and planting age. We further 260 discuss these findings in the remainder of this paper and conclude with some commentary of 261 their implications for restoration management and biodiversity conservation. 262

263 Grazing effects

The most prevalent grazing effect identified in our study was an indirect one in which grazing (and particularly uncontrolled grazing), reduced the amount of leaf litter. This, in turn, lead to depressed bird species richness, reduced reptile species richness, and reduced occurrence of the White-plumed Honeyeater and Superb Fairy-wren (Figs. 2, 3, S2, S3). Earlier studies have shown that livestock grazing leads to a significant loss of leaf litter, albeit in temperate woodland remnants rather than plantings (e.g. Robertson & Rowling 2000; Yates et al. 2000). Leaf litter is, in turn, an important foraging substrate for many woodland
birds (Antos et al. 2008; Barrett et al. 2008) and also reptiles (Valentine et al. 2007). Grazingrelated modification of the suitability of habitat and foraging substrate suitability is therefore
a plausible explanation for the indirect pathway linking grazing and altered bird and reptile
occurrence.

We also uncovered evidence of direct effects of grazing not mediated through 275 modification in vegetation condition. Such effects may manifest through trampling and 276 perturbation of the ground and affect nests, for example). Such a mechanism is plausible for 277 278 species such as the Rufous Songlark which nests on the ground. This effect is consistent with work by (Ford 2011) who recognized that ground foraging and ground nesting woodland 279 birds were prominent among those declining across temperate woodland biomes in south-280 281 eastern Australia. However, trampling-related perturbation may not explain the direct effects of grazing on the White-plumed Honeyeater, the abundance of which was reduced depending 282 on whether plantings were grazed or not (Table S9, Fig. S2). The White-plumed Honeyeater 283 only infrequently forages on the ground and other factors may be important for this species. It 284 may nest as low as 1 meter above the ground and mechanical disturbance by livestock might 285 influence nesting success. The White-plumed Honeyeater also uses spider web to construct 286 its nests (Higgins et al. 2001) and work elsewhere in south-eastern Australia has shown that 287 abundant (albeit native) populations of herbivores can cause significant mechanical damage 288 289 on the webs constructed by spiders (Foster et al. 2015).

290 Planting width effects

We uncovered evidence of a direct positive effect of planting width on bird species richness (Fig. 2). This suggests that geometry is important for birds in restored environments; for example, landscape ecology theory predicts that wider plantings will have more interior versus edge habitat suitable for animal occupancy of sites (Lindenmayer & Hobbs 2007;Collinge 2009).

An unexpected outcome of our path analysis was that wider plantings were 296 297 characterized by vegetation with lower values for midstory cover. The reasons for this relationship remain unclear but such paths had subsequent links with reduced overall bird 298 species richness (Fig. 2) and, conversely an increased occurrence of the Rufous Songlark 299 (Fig. S1). Midstory vegetation can add an important layer in the vertical structure of a stand 300 of woodland and can, in turn, add to the number of niches available for different species of 301 302 birds - the vegetation structure hypothesis underpinning overall bird species richness (MacArthur & MacArthur 1961). However, some ground-associated species may be 303 304 disadvantaged by additional layers of vegetation cover as observed for the occurrence of the 305 Rufous Songlark (Fig. S1).

306 Planting age

Only one species, Boulenger's Skink, exhibited a direct positive response to the age 307 of plantings. That is, the species was more likely to occur in older plantings. One explanation 308 for this was that the longer plantings have been established, the greater the amount of natural 309 self-thinning of trees and, in turn, the more light penetrating to the woodland floor, thereby 310 creating more suitable habitat for this generalist reptile species. However, we found no 311 312 indirect relationships between planting age, vegetation structure and condition and bird or 313 reptile response. This result was unexpected given that vegetation structure and composition of planted areas changes over time with vegetation succession and maturation (e.g. Vesk et 314 al. 2008) and this can influence the availability of potential food sources such as invertebrate 315 316 prey (Majer et al. 2001; Gibb & Cunningham 2010) as well as the abundance of flowers, pollen, nectar and seeds. Although our plantings were between six and > 20 years old at the 317 start of our investigation, it may be that more time is required for the ecological maturation of 318

restored areas (Munro et al. 2009) and, in turn, the emergence of indirect planting effects onbirds and reptiles.

321 Management implications

322 This study and several previous investigations by us (Lindenmayer et al. 2007; Lindenmayer et al. 2010b; Munro et al. 2011; Pulsford et al. 2017) and other researchers (e.g. 323 Ryan 2000; Robinson 2006; Barrett et al. 2008; Kinross & Nicol 2008; Selwood et al. 2008) 324 have highlighted the value of revegetated areas for biodiversity in the temperate woodland 325 environment of eastern Australia. The study we report here indicates that the benefits of 326 327 restored areas may be undermined when they are grazed, especially by uncontrolled grazing. These findings and others from this investigation have important implications for the 328 management of restored areas. 329

330 First, our results suggest that where the objective of vegetation restoration is to enhance biodiversity conservation, it may be appropriate to limit the amount of grazing 331 within established plantings. This recommendation has, in turn, important ramifications for 332 the maintenance of key infrastructure such as fencing as this is currently the primary method 333 to control the intensity, frequency and periodicity of livestock grazing on farms (Spooner & 334 Briggs 2008). Indeed, our work showing the negative effects of grazing on bird and reptile 335 species richness is timely given that 20 or more years after many plantings were established 336 within (but also beyond) our study, fences are deteriorating as a result of natural attrition and 337 338 need to be replaced or subject to substantial maintenance.

A second important implication from our study concerns the width of plantings. We suggest that where possible, wide plantings should be established given that such areas support higher levels of bird species richness (see also Kinross 2004; Munro et al. 2007). Many past restoration programs in our study region have resulted in the creation of narrow

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strips of planted woodland. These areas are not without value, but we suggest that there canbe greater biodiversity gains if plantings are widened when fences need to be replaced.

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489 **Figure captions**

- 490 Figure 1. Conceptual model (path diagram) of potential inter-relationships between
- 491 management, vegetation characteristics and biodiversity response in Australian temperate
- 492 woodlands.



494

Figure 2. Directed acyclic graph (DAG) depicting which planting attributes, grazing and
vegetation variables are important in the path analysis of bird species richness. Path
coefficients are deemed important if their 95% credible intervals do not overlap zero (see
Supplementary Information Table S4). Coefficients whose credible intervals overlap zero or
were excluded from the final model via LOOIC model selection are omitted from the path
diagram. The larger the absolute value of the coefficient, the stronger the effect.



Figure 3. Directed acyclic graph (DAG) depicting which planting attributes, grazing and
vegetation variables are important in the path analysis of reptile species richness. Path
coefficients are deemed important if their 95% credible intervals do not overlap zero (see
Supplementary Information Table S4). Coefficients whose credible intervals overlap zero or
were excluded from the final model via LOOIC model selection are omitted from the path
diagram. The larger the absolute value of the coefficient, the stronger the effect.

