Evaluating complementary networks of restoration plantings for landscape-scale occurrence of temporally-dynamic species

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1 **Abstract**

2 Multi-billion dollar investments in land restoration make it critical that conservation goals are achieved cost-

3 effectively. Approaches developed for systematic conservation planning offer opportunities to evaluate

4 landscape-scale, temporally-dynamic biodiversity outcomes from restoration, improving on traditional

5 approaches of focusing on the most species-rich plantings. We investigated whether it is possible to apply a

6 complementarity-based approach to evaluate the extent to which an existing network of restoration plantings

7 meets representation targets. Using a case study of woodland birds of conservation concern in southeastern

8 Australia, we compared complementarity-based selections of plantings based on temporally-dynamic species

9 occurrences with (1) selections based on static species occurrences, and (2) selections based on ranking

10 plantings by species richness. We found that incorporating dynamics in species occurrences across a five-year

11 period resulted in higher species occurrences and proportion of targets met compared with using species

12 occurrences taken at a single point in time. For equivalent cost, the dynamic complementarity approach also

13 always resulted in higher average minimum percent occurrence of species maintained through time and a

14 higher proportion of the bird community meeting representation targets compared with the species richness

15 approach. Plantings selected under the complementarity approaches represented the full range of planting

16 attributes, whilst those selected under the species richness approach were larger in size. Future restoration

17 policy should not attempt to achieve all conservation goals within individual plantings, but should instead

18 capitalise on restoration opportunities as they arise to achieve collective value of multiple plantings across the

19 landscape. Networks of restoration plantings with complementary attributes of age, size, vegetation structure,

20 and landscape context lead to considerably better outcomes than conventional restoration objectives of site-

21 scale species richness, and are crucial for allocating restoration investment wisely to reach desired

22 conservation goals.

23

24

Introduction

Restoration plantings are a widely implemented biodiversity conservation approach in agricultural landscapes (Bullock et al. 2011). Worldwide, international biodiversity targets for 2020 include the restoration of at least 15% of degraded ecosystems (Aichi Biodiversity Target 15, CBD COP 2010). Similar international targets for the restoration of 150 million ha of degraded lands by 2020 (Rio+20, UNCSD 2012) are estimated to cost nations globally US$18 billion per year (Menz et al. 2013). Specific restoration targets have been set by the European Union (Bullock et al. 2011) and individual countries, including Australia, whose Biodiversity Fund aims to restore 18 million ha of native vegetation by 2020 with a budget of US$1 billion (Australian Government 2013). Given this substantial investment, it is important that restoration initiatives are both efficient and cost-effective, maximising biodiversity outcomes for the least cost (Menz et al. 2013).

Most research on biodiversity outcomes within restoration plantings (a form of active restoration) has focused on whether and how individual plantings achieve high species occurrence, richness or abundance (Munro et al. 2007). In addition to comparing the value of restoration plantings to that of reference sites (e.g. Gould et al. 2013), these studies have identified attributes of plantings that contribute to increased biodiversity at the site-scale, including planting age (Vesk et al. 2008), area and shape (Lindenmayer et al. 2010; Jellinek et al. 2014) and vegetation structure (Munro et al. 2011). This earlier research recommends that future restoration investment be focused on maximising site-scale attributes related to high individual planting biodiversity. Restoration plantings, however, also have value at the landscape scale (Thomson et al. 2009; Rappaport et al. 2015), and the collective features of different plantings across the landscape may be a better measure of biodiversity value than site-scale attributes. An alternative approach to restoration investment, therefore, is to maximise the number of species present across the entire landscape through a focus on networks of restoration plantings.

Systematic conservation planning, originally developed for locating and designing cost-effective protected areas (Margules & Pressey 2000), has increasingly been used to spatially prioritise new areas for restoration (Thomson et al. 2009; Lethbridge et al. 2010; McBride et al. 2010; Wilson et al. 2011; Yoshioka et al. 2014; Possingham et al. 2015). A key concept is complementarity, which ensures that each restoration planting

53   contributes unrepresented features to the larger network of plantings (i.e. that each planting complements the

54   others in the network) (Margules & Pressey 2000). Complementarity approaches to the selection of restoration

55   plantings thus differ from selection based on traditional measures of conservation value that focus on the most

56   species-rich plantings. This is because plantings with high individual species richness may not necessarily

57   contribute to overall conservation goals of maximising diversity at a landscape or regional scale (Margules &

58   Pressey 2000; see also Chadès et al. 2015). Systematic conservation planning has frequently been used to

59   evaluate the performance of an existing set of protected areas (e.g. Stewart et al. 2003), and the same approach

60   might be useful to evaluate the performance of an existing network of restoration plantings. Undertaking such

61   an evaluation would: (a) identify the best complementary subset of plantings that contribute the most to the

62   biodiversity benefits of the network, and might be afforded protection in cases of impact assessment and

63   future landscape clearing, and (b) elucidate the attributes of plantings important for landscape-scale

64   biodiversity outcomes. In doing so, landscapes undergoing restoration might have more efficient investment

65   and conservation outcomes.

66

67   Most systematic conservation planning usually considers species occurrence only at a single point in time, but

68   plant and animal communities (particularly those in disturbed landscapes) are temporally dynamic (Grantham

69   et al. 2011; Runge et al. 2014; Tulloch et al. 2016). Failure to incorporate dynamics into spatial prioritisations

70   – e.g. using static species distributions derived from a single year of data or pooled over years – can lead to

71   insufficient representation of species over time (Runge et al. 2016; Tulloch et al. 2016). Whist previous spatial

72   prioritisations for restoration have considered dynamics in the age and structural complexity of restored

73   vegetation (e.g. Thomson et al. 2009), we are not aware of any spatial prioritisation study that has accounted

74   for temporal dynamics in the distribution or occurrence of species colonising restoration plantings, potentially

75   undermining the success of restoration schemes.

76

77   In this study, we investigated whether taking a temporally-dynamic complementarity approach can evaluate

78   the contribution of existing restoration plantings to achieve landscape-scale species occurrence. We used, as a

79   case study, a network of plantings in the South West Slopes Bioregion of southeastern Australia. Only 15% of

80   the once extensive temperate eucalypt woodland remains within this agricultural region (Benson 2008) and,

81 consequently, many woodland bird species are of conservation concern (Rayner et al. 2014). Since 1990,

82 however, concerted investment has been made in establishing restoration plantings for a range of conservation

83 and land management objectives, including increasing woodland bird habitat. Through extensive programs

84 managed by multiple stakeholders, thousands of hectares of vegetation have been planted, corresponding to

85 increases of 3-4% of vegetation cover in the landscape (Lindenmayer et al. 2012; Cunningham et al. 2014). As

86 part of the South West Slopes Restoration Study (Cunningham et al. 2007), 61 plantings have been surveyed

87 for birds and vegetation in five spring seasons since 2006.

88

89 Our first aim in this study was to find the best complementary network (i.e. subset) of established restoration

90 plantings to support landscape-scale occurrence of species of conservation concern for minimal establishment

91 cost. We accounted for temporally dynamic species occurrences, by requiring representation targets for

92 species occurrence to be met in every year (Runge et al. 2016). We compared the outcomes of taking a

93 dynamic complementarity approach to find a network of plantings that met our desired representation target

94 with (1) networks selected using a static complementarity approach based on single years of data, and (2)

95 equivalently costed networks of plantings ranked by richness of species of conservation concern.

96

97 Our second aim was to identify the attributes of plantings that contributed most to the landscape-scale

98 occurrence of species of conservation concern. The plantings in our study were established for a variety of

99 reasons (e.g. wind breaks, soil erosion and salinity), varied in age, area, shape, vegetation structure, and

100 landscape context, and subsequently varied in their individual value for woodland birds (Lindenmayer et al.

101 2010). This opportunistically created a network of a plantings that was ideal for exploring how subsets of

102 plantings with different characteristics differed in terms of their ability to represent all bird species of

103 conservation concern. Our study thus quantifies the value of evaluating biodiversity benefits of management

104 at the landscape scale and incorporating temporally dynamic species distributions into restoration planning.

105 The work informs future investment to ensure more efficient and cost-effective biodiversity outcomes across

106 restoration landscapes.

107

Methods

Study area, experimental design, and data collection

108 **Methods**

109 *Study area, experimental design, and data collection*

110 The South-West Slopes Restoration Study is located in a 150 km x 120 km area of the South West Slopes

111 Bioregion of New South Wales, Australia (Fig. 1). This region was once dominated by temperate box-gum

112 *Eucalyptus* woodland, but is now characterised by cropping and livestock grazing. Farms typically have

113 between 3% and 35% native vegetation cover, including old growth woodland, regrowth, and plantings

114 (Cunningham et al. 2014). For this investigation, we used data from 61 plantings, distributed across 25 farms.

115 These plantings were not established following any plant species composition or spacing protocols but,

116 typically, planting composition was a mix of local endemic and exotic Australian ground cover, understorey

117 and overstorey species, with plants spaced ~2 m apart. For each planting, we compiled data on variables

118 known to be important for bird richness and occurrence in restoration plantings: years since establishment,

119 area and width of plantings, vegetation structural complexity, surrounding woody vegetation cover (a proxy

120 for connectivity), and landscape position (see Table S1 and S2 in the Supporting Information).

121

122 We used the area and shape of plantings to estimate the total establishment cost of each planting. Our

123 estimates were based on 2015 pricing rates used by Greening Australia, one of Australia's largest and longest-

124 running restoration practitioners. We calculated costs ($AUD) of materials and labour for fencing and direct-

125 seeding of sites (Table S3). As our focus was on biodiversity as a public benefit, we considered only public

126 costs of establishing restoration sites in this analysis. We acknowledge the importance of considering private

127 opportunity costs and ongoing management costs in conservation planning on public land, but the inclusion of

128 such information was beyond the scope of this study.

129

130 We collected bird occurrence data in the spring seasons of 2006, 2008, 2009, 2011 and 2013. In each year,

131 every planting was visited twice within a four day period in early November, on two days by different

132 observers, and a five-minute point count was conducted at the 0 m, 100 m and 200 m points of a permanent

133 transect. All birds seen or heard within 50 m of the point, excluding those flying overhead, were recorded as

134 present. Surveys were conducted between sunrise and mid-morning, avoiding inclement weather. This strict

135 survey protocol is designed to address biases in observer heterogeneity (Cunningham et al. 1999) and false-

136   negative errors, i.e. failure to detect species that are present (Banks-Leite et al. 2014), and is applied across all

137   our long-term major studies in woodlands.

138

139   We defined woodland birds of conservation concern as those species: (i) dependent on woodland for foraging

140   and/or nesting (Silcocks et al. 2005), and (ii) listed as threatened in New South Wales under the *Threatened*

141   *Species Conservation Act 1995* (this also captures relevant nationally-listed threatened species) and/or

142   identified as having a >20% decrease in South West Slopes bioregion reporting rate between the first and

143   second Atlas of Australian Birds (Barrett et al. 2003). Excluding very rare species (only recorded once during

144   the five years), this definition resulted in 26 species of conservation concern for analysis (Table S4). We used

145   Permutational Multivariate Analysis of Variance (PERMANOVA) to test for significant differences in species

146   composition between years, based on a Bray-Curtis dissimilarity matrix adjusted for species presence/absence

147   data, using the *vegan* package in R (R Development Core Team 2007).

148

149   *Data analysis*

150   To address our first aim (i.e. achieve targets of landscape-scale occurrence of species of conservation concern

151   for least investment), we compared the outcomes of using 'dynamic' versus 'static' complementarity

152   approaches, and 'complementarity' versus 'ranked' approaches, to find the best subset network of restoration

153   plantings. We set an objective of representing targets of 10% to 100 % (10% increments) occurrence of each

154   species per year in all years (equivalent to 10% to 100% of plantings where each species occurred in each

155   year).

156

157   To find the best complementarity-based networks of plantings for each target, we used the decision-support

158   software, Marxan, which uses a simulated annealing algorithm to solve the minimum set problem (Ball et al.

159   2009). The objective was to minimise resources expended (i.e. cost of the planting network) whilst meeting

160   pre-specified representation targets (i.e. scenarios of 10% to 100% individual species occurrence per year in

161   all years). To account for temporal variation in species occurrence between plantings (planning units), we

162   created a conservation feature for each species for each survey year (five conservation features per species of

163   conservation concern, giving 130 conservation features in total), following Runge et al. (2016).

164   Representation of conservation features in a given planting was based on presence/absence data, i.e. whether

165   or not each species was recorded in each planting in each year. For each increasing 10% target scenario, we

166   compared the 'dynamic' approach with five 'static' approaches, based on single years of data (2006 only,

167   2008 only, 2009 only, 2011 only, and 2013 only). The objective of the static approaches was to meet

168   representation targets only for that particular year. We parameterised Marxan to find the most cost-effective

169   network irrespective of spatial configuration (by setting the boundary length modifier to zero), and performed

170   100 runs per scenario. We confirmed that the selected networks were not driven by planting cost by

171   comparing the scenarios with baseline 'no cost' scenarios (Table S5). We considered two Marxan outputs for

172   each scenario: the 'best' network of plantings that met the representation target for the least cost, and the

173   'selection frequency' (i.e. irreplaceability) of each planting (the number of times each planting was selected

174   across the 100 runs). For our scenarios, these two values were strongly positively correlated (≥ 0.9), and the

175   average selection frequency for plantings selected in the best network was close to 100% (Table S6). Because

176   this indicates high irreplaceability in selected plantings, we used the 'best' networks of plantings for

177   subsequent analyses.

178

179   We paired each dynamic complementarity scenario with a ranked scenario of equivalent cost, creating ten

180   matched pairs of networks (i.e. one for each species occurrence target (10% to 100%)). To do this, we

181   calculated total richness of species of conservation concern across the five survey years and ranked individual

182   plantings from high to low species richness. We calculated the cumulative cost of the plantings based on these

183   rankings, and included in the best network only those plantings that could be afforded for less than or equal to

184   the cost of the dynamic complementarity scenario.

185

186   For each network selected by the dynamic, static, and ranked approaches, we calculated the: (i) cumulative

187   establishment cost, (ii) number of plantings in the network, and (iii) summary statistics for the minimum

188   percentage of the occurrence of each species that was met over the five years. We also calculated Bray-Curtis

189   dissimilarity (adjusted for presence/absence data) between networks to assess spatial concordance between the

190   selected plantings (e.g. low Bray-Curtis dissimilarity between a pair of networks indicates that the spatial

191   locations of the plantings in the networks were similar). We confirmed that differences between the dynamic

192 complementarity and ranked networks were not driven by cost-effectiveness by comparing our results with

193 equivalently-costed networks that were based on ranking plantings by cost-effectiveness but ignoring

194 complementarity (dividing species richness by cost) (Table S8).

195

196 To address our second aim (i.e. identify the attributes of plantings that contribute to landscape-scale

197 occurrence of species of conservation concern), we modelled the relationship between planting attributes and

198 the probability of the planting being selected in the dynamic and static complementarity and ranked scenarios

199 for two representation targets (30% and 60% species occurrence in all years). We also modelled the number of

200 times (frequency) each planting was selected in the static networks for these targets over the five years. The

201 first target (30% occurrence) was chosen to reflect typical targets for conservation assessments (Svancara et

202 al. 2005). The second target (60% occurrence) was chosen based on the results of the Marxan analyses, as

203 there was a threshold jump in planting benefits at this target level for the dynamic complementarity approach.

204 Planting attributes included standardised site-level variables (Table S1). Planting width was strongly and

205 positively correlated with planting area, and so was excluded from further analyses We adopted an

206 information theoretic approach to model selection (Burnham & Anderson 2002), and compared a candidate set

207 of 31 models that included single and additive combinations of all planting attributes (Table 2). We

208 considered the univariate planting area model to be the null model, as previous research suggests that this

209 attribute is of primary importance in restoration (e.g. Lindenmayer et al. 2010). We fitted generalised linear

210 models (GLM), with a binomial error distribution and log link (*AICcmodavg* package).We modelled each

211 response variable against a distance-weighted spatial autocovariate (*spdep* package) to check for spatial

212 autocorrelation between sites. For response variables that showed evidence of spatial autocorrelation, we

213 included the distance-weighted spatial autocovariate in each alternative model. We ranked the candidate set of

214 models by Akaike's Information Criterion corrected for small sample bias (AICc). For top-ranked models

215 (within 2 ΔAICc of the model with the lowest AICc), we assessed model support using Nagelkerke's

216 coefficient of determination (R^2; *fmsb* package), and calculated model-averaged effect sizes for the model

217 terms.

218

219 **Results**

220 Over the five survey years, we recorded 100 woodland bird species, including 26 of conservation concern

221 (Table S4). Total richness of species of conservation concern ranged from 1 to 14 species per planting.

222 Species composition differed significantly between years ($F = 2.723$, $p = 0.006$).

223

224 *Dynamic vs static complementary restoration planting networks*

225 The complementarity approach that incorporated dynamic species occurrences consistently resulted in higher

226 mean minimum percent occurrence of species across the five survey years than the static complementarity

227 approaches based on single years of data (Fig. 2a, Table S7). Although more expensive to achieve any given

228 target than the static approaches, the dynamic approach always met the representation target for every species

229 (Fig. 2b). In comparison, although plantings selected using a static single-year approach met the

230 representation target for that year, they failed to meet the representation target across all time (2006-2013) for

231 more than a third of species. This is because all 61 plantings were required to meet the dynamic representation

232 target of 100% occurrence for each species across all time, compared with between 42 and 54 plantings for the

233 static targets.

234

235 The spatial locations of the best network of selected plantings differed markedly between years. For example,

236 for the 30% target, there was 44 % – 78 % Bray-Curtis dissimilarity in selected plantings between years

237 (Table 1). To meet this target, each planting was selected an average of 1.46 times (out of 5 possible static

238 networks); 20 plantings were never selected and only one planting was always selected. The spatial locations

239 of the selected plantings also differed between the dynamic and static approaches (average Bray-Curtis

240 dissimilarity for the 30% target = 49%). However, within each approach, plantings selected under low

241 representation targets were usually also selected under higher targets (average Bray-Curtis dissimilarity

242 between increments = 12%).

243

244 *Dynamic complementary vs ranked restoration planting networks*

245 The dynamic-complementarity approach consistently resulted in higher mean percent species occurrence than

246 the species-richness ranked approach (Fig. 3a, Table S8). For equivalent cost, mean minimum percent

10

247   occurrence of species was up to 30% higher in the complementarity scenarios. Further, whilst the

248   representation target was achieved in every complementarity scenario (i.e. all species met the specified target),

249   up to 46% of species did not meet the target in the equivalent-costed ranked scenarios (Fig. 3b). On average,

250   there was 78% overlap in the spatial location of plantings selected under the complementarity and ranked

251   approaches (average Bray-Curtis dissimilarity = 22%).

252

253   *Relationships with planting attributes*

254   Plantings selected under the static and dynamic complementarity approach for the 30% and 60% targets did

255   not consistently differ in their attributes from the non-selected plantings. Model uncertainty was high, as top-

256   ranked models had relatively low R^2 values (Table 2). The effect sizes of terms in the models were generally

257   small and variable (confidence intervals crossed 0) (Fig. 4a – c, Fig. S1). Similarly, there was no consistent

258   relationship between the number of times each planting was selected in the static networks over the five years

259   and planting attributes (Fig. 4d). Plantings selected more frequently to meet the 30% target were younger and

260   surrounded by more woody vegetation cover but effect sizes were small, and these effects were variable for

261   plantings selected to meet the 60% target. In comparison, plantings selected under the richness ranked

262   scenarios were larger than non-selected plantings, and model certainty was relatively high (Table 2, Fig. 4e).

263

264   **Discussion**

265   The restoration of degraded lands is an international conservation goal, with multi-billion dollar annual

266   investment, requiring wise allocation of resources (Bullock et al. 2011; Menz et al. 2013). Our study shows

267   that it is possible to apply the principles of systematic conservation planning to evaluate the extent to which an

268   existing network of restoration plantings meets representation targets for woodland birds of conservation

269   concern. Incorporating dynamics in species occurrences across a five-year period resulted in higher species

270   occurrences and proportion of targets met compared with using species occurrences representing a single

271   point in time. Importantly, we found that for equivalent cost, the dynamic complementarity approach always

272   resulted in higher average minimum percent occurrence of species maintained through time and a higher

273   proportion of the bird community meeting representation targets compared with ranking plantings by species

274   richness (Aim 1). We also found that plantings selected to achieve goals of both representation and

275   complementarity represented the full range of planting attributes, whilst those selected under the richness

276   approach were larger in size (Aim 2).

277

278   We found that incorporating dynamic species occurrences led to more expensive networks of restoration

279   plantings, but considerably higher long-term species occurrences and achievement of representation targets

280   compared with static approaches. This was because the bird community was highly spatially and temporally

281   dynamic, with little overlap between networks selected based on single years of data. Compared with static

282   distribution approaches, incorporating temporally-dynamic species ranges in systematic conservation planning

283   leads to more expensive and less flexible networks, but improved biodiversity outcomes (see also Grantham et

284   al. 2011; Lourival et al. 2011; Van Teeffelen et al. 2012). For example, Runge et al. (2016), found that

285   accounting for annual and seasonal range variation in nomadic bird species led to greater areas of land

286   needing to be conserved to achieve targets but greater levels of species protection. Similarly, in their case

287   study of the South American Pantanal wetlands, Lourival et al. (2011) found that incorporating dynamic

288   vegetation distributions, although increasing expense, improved the reliability and long-term adequacy of their

289   reserve networks. A dynamic prioritisation approach is thus crucial for allocating investment wisely to reach

290   desired conservation goals (Tulloch et al. 2016).

291

292   Using a landscape-scale complementarity approach was critical to achieve cost-effective subsets of restoration

293   plantings across the existing network. For example, to achieve similar species representation (for targets

294   ≤90% species occurrence), the complementarity approach required less investment, fewer plantings, and less

295   combined restored area than the ranked approach based on site-scale species richness. Further, even with the

296   substantial additional investment needed for the ranked approaches, many species still did not meet the

297   representation target in every year (compared with all targets achieved under the complementarity approach).

298   Complementarity approaches to reserve design have long been recognised as superior to ranked approaches

299   (Chadès et al. 2015), and our study supports their utility in restoration programs (Yoshioka et al. 2014).

300   However, by definition, the high efficiency that complementarity achieves may result in low redundancy

301   across the network of restoration sites, with implications for network robustness to disturbance (O'Hanley et

302   al. 2007). In our scenarios, we incorporated multiple years of data, including from severe drought (2006 -

303 2009) and post-drought recovery (2011 - 2013). Incorporating these dynamics within our system likely

304 reduced the trade-off between complementarity and robustness through accounting for stochastic processes

305 (Lourival et al. 2011; Van Teeffelen et al. 2012).

306

307 Our finding that no single attribute makes plantings best for bird occupancy over space and time challenges

308 conventional thinking that there is a type of restoration planting best for woodland birds (Lindenmayer et al.

309 2010). Instead, our findings support previous research on the differing and complementary habitat suitability

310 of plantings for different functional groups (Loyn et al. 2007). By collectively considering occupancy of

311 plantings by each species in our analyses, we were able to specifically account for the variable habitat

312 requirements of our bird community. However, it is difficult to evaluate to what extent the bird occurrence

313 patterns within the best networks of plantings were influenced by bird occurrence in unselected plantings (to

314 which highly-mobile taxa like birds could disperse) or by other vegetation types (e.g. regrowth and remnant

315 vegetation) in the study landscape (Lindenmayer et al. 2012). Future research should investigate

316 complementarity and connectivity between restoration plantings, regrowth vegetation (i.e. passive

317 restoration), and remnant vegetation for landscape-scale species persistence. Future research could also

318 integrate dynamics in planting attributes with dynamics in species occurrences. For instance, in our study we

319 held planting attributes constant, yet some attributes such as age, structural complexity, and connectivity may

320 change through time (Thomson et al. 2009). As such, the attributes of plantings that are likely to maximise

321 complementarity may also change through time, as suggested by our findings from our static models. Ideally,

322 any future research that uses cost-effectiveness analysis to prioritise restored habitat in agricultural landscapes

323 should also incorporate costs associated with lost farming opportunities in restored areas (Naidoo et al. 2006).

324

325 Translating our findings into future restoration policy involves some challenges. Our results show that it is

326 desirable from a complementarity perspective to encourage a mixed portfolio of restoration projects that differ

327 in the attributes of plantings and landscape context. Given real-world social, economic and political

328 constraints on biologically-driven conservation planning 'informed opportunism' (sensu Noss et al. 2002) may

329 be appropriate. That is, in addition to available biodiversity knowledge, future investment in restoration

330 initiatives also should be guided by the capacity and willingness of land owners to participate (Knight et al.

331 2010). A key difficulty is developing policy that can capitalise on informed opportunism to achieve

332 complementary planting networks. One approach may be to implement policies that support consistent,

333 incremental funding of restoration plantings in a region, so that a breadth of planting ages and structural

334 attributes is maintained. Another more resource-intensive approach could be to allocate funding for new

335 plantings that would complement the attributes of existing restoration plantings.

336

337 In summary, our study shows that a complementarity approach can be used to find the best network of

338 established restoration plantings, and that this network is more cost-effective and represents more of species'

339 landscape occupancy than a traditional species richness approach,. Further, incorporating temporally-dynamic

340 species occurrences leads to a more cost-effective and robust restoration plantings network compared with

341 using static single-year data (Grantham et al. 2011; Lourival et al. 2011; Van Teeffelen et al. 2012; Runge et

342 al. 2014). Substantial resources will continue to be invested in restoration initiatives in response to

343 international and national policy, and as part of wider agri-environmental schemes (Bullock et al. 2011; Menz

344 et al. 2013). This investment should not attempt to achieve all conservation goals within individual plantings,

345 but could instead be implemented incrementally to capitalise on restoration opportunities as they arise (Noss

346 et al. 2002) to achieve collective value of multiple plantings across the landscape. Adopting a landscape-scale

347 temporally-dynamic approach leads to considerably better outcomes for a faunal community of conservation

348 concern than applying conventional site-scale metrics, and is crucial for the wise allocation of restoration

349 investment to reach desired conservation goals.

350

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Tables and Figures

Table 1: Percent Bray-Curtis dissimilarity between the spatial locations of plantings selected in the static and dynamic complementarity restoration planting networks for the 30% representation target.

	2006	2008	2009	2011	2013
2008	44.44				
2009	52.94	57.89			
2011	54.29	48.72	78.38		
2013	43.75	50.00	58.82	54.29	
Dynamic	50.00	38.46	36.00	45.10	41.67

Table 2: Candidate set of models. Nagelkerke's coefficient of determination (R^2) is presented for the top-ranked models (ΔAICc [a] ≤ 2). 'Area' is planting area, 'Age' is years since planting establishment, 'HCS' (habitat complexity score) represents vegetation structural complexity, 'Woody Veg' is percentage of vegetation cover within 1 km, and 'TWI' (topographic wetness index) represents position in landscape ranging from ridges to valley floors.

Model	2006		2008		2009		2011		2013		Dynamic		Frequency [b]		Ranked	
	30%	60%	30%	60%	30%	60%	30%	60%	30%	60%	30%	60%	30%	60%	30%	60%
Area [c]	*0.08*	0.04	*0.00*	*0.01*	0.01	0.00	*0.04*	*0.02*	*0.01*	*0.01*	0.05	0.06	*0.17*	*0.07*	0.20	0.27
Age			0.20		0.02	0.02	0.18	0.07	0.08	0.10	0.01	0.04		0.29		
Age + Woody Veg			0.15	0.21	0.02		0.23	0.10		0.11	0.07	0.10	0.36	0.31		
Area + Age	0.14				0.04		0.19	0.13	0.10	0.14		0.08				0.27
TWI		0.04			0.01	0.00	0.06	0.09			0.03	0.03				
Age + TWI				0.21	0.02		0.20	0.11	0.14	0.13				0.31		
Age + Woody Veg + TWI							0.26	0.16		0.16	0.11	0.15	0.40	0.35		
Woody Veg		0.04	0.09		0.00	0.00					0.05	0.06				
Area + TWI		0.10						0.10			0.10	0.12			0.21	0.28
Area + Woody Veg	0.15	0.12									0.08	0.09			0.21	
Woody Veg + TWI			0.11					0.11			0.10	0.12				
Area + Age + Woody Veg	0.22		0.16							0.18			0.40			
Area + Age + TWI							0.14	0.15	0.16			0.13				

Model													
Area + Woody Veg + TWI		0.16				0.18			0.14	0.17			
Area + HCS					0.03							0.21	0.28
Age + HCS					0.03	0.20	0.12						
Age + HCS + Woody Veg			0.15			0.23	0.10						
Area + Age + Woody Veg + TWI								0.20		0.18	0.43		
HCS				0.00	0.02								
Area + HCS + Woody Veg	0.20	0.17											
Age + HCS + TWI						0.15	0.16						
HCS + Woody Veg			0.12										
HCS + TWI							0.13						
HCS + Woody Veg + TWI									0.11				
Area + Age + HCS + Woody Veg	0.23												
Age + HCS + Woody Veg + TWI						0.14							
Area + Age + HCS + Woody Veg + TWI											0.44		
Area + Age + HCS [d]													
Area + Age + Woody Veg [d]													
Area + HCS + TWI [d]													
Area + Age + HCS + TWI [d]													

[a] delta Akaike Information Criterion adjusted for small sample size. [b] Spatial autocovariate included in all alternative models. [c] Null model. Values in italics indicate that the model was not within the top-ranked model set. [d] Not a top-ranked model.
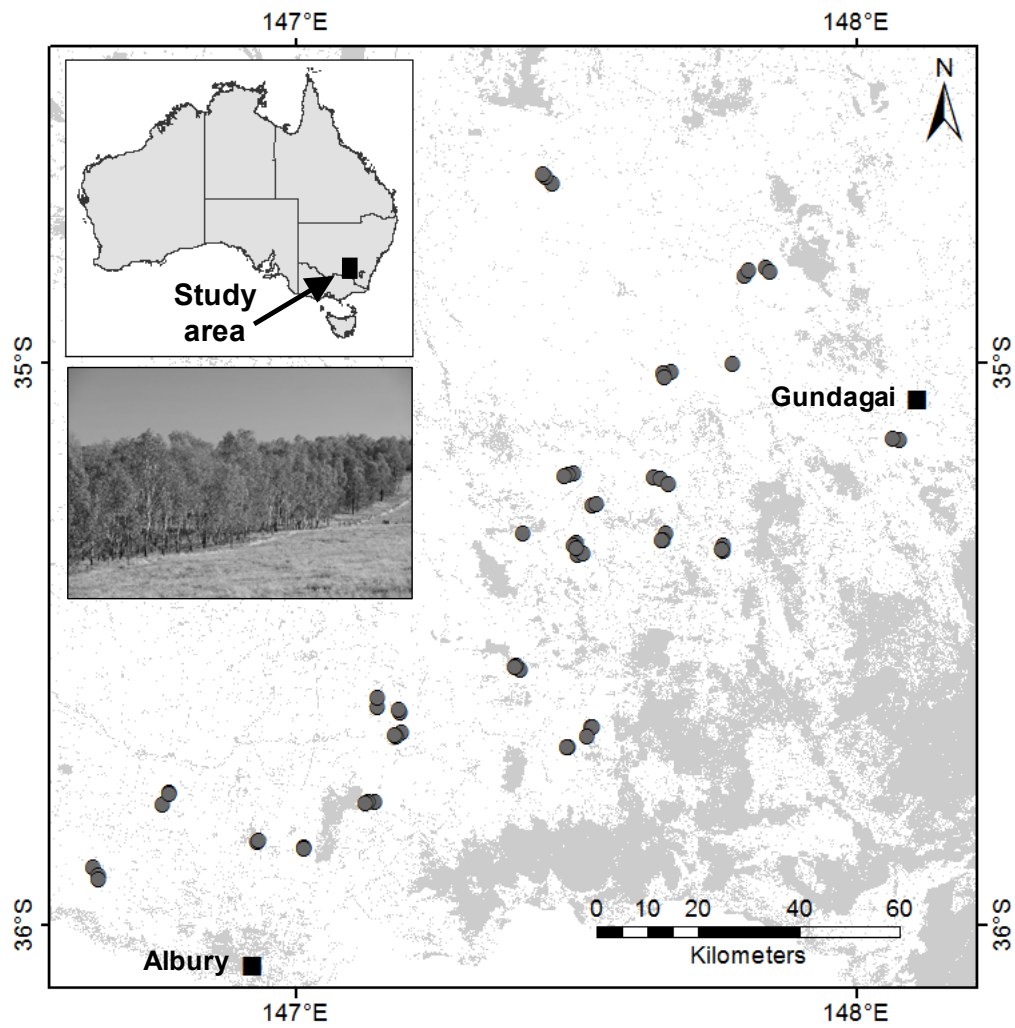
Figure 1: Map of study area showing restoration planting sites (note that points are not drawn to scale). Grey shading shows native woody vegetation cover. Insets show location of study area within Australia (top) and image of a typical planting site (bottom). Image by D. Blair.
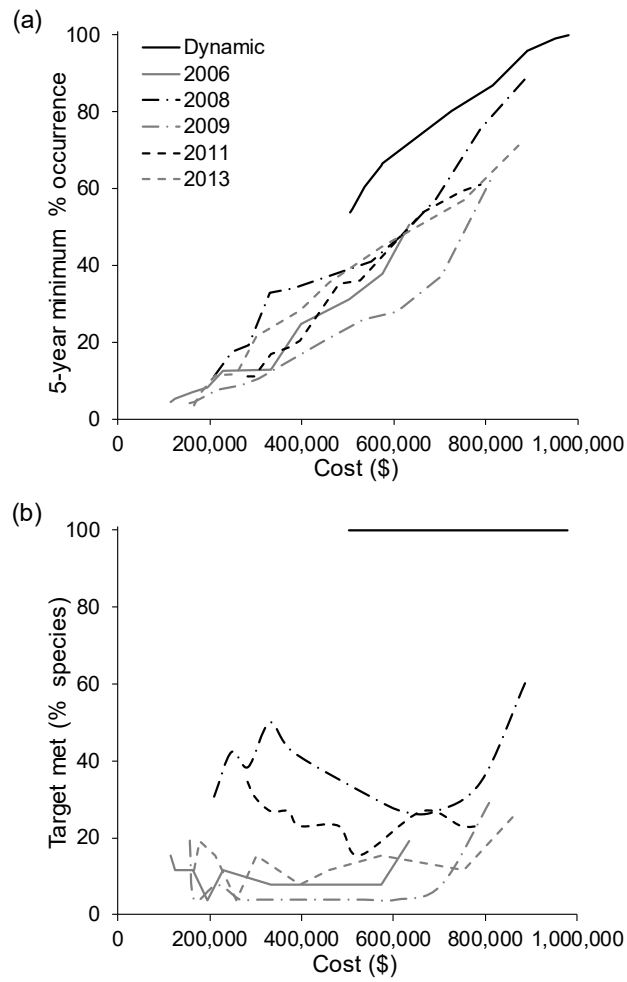
Figure 2. Comparison of five-year outcomes under dynamic and static complementarity approaches for (a) 5-year mean minimum percent occurrence, and (b) targets met. Each line represents 10% to 100% representation targets.
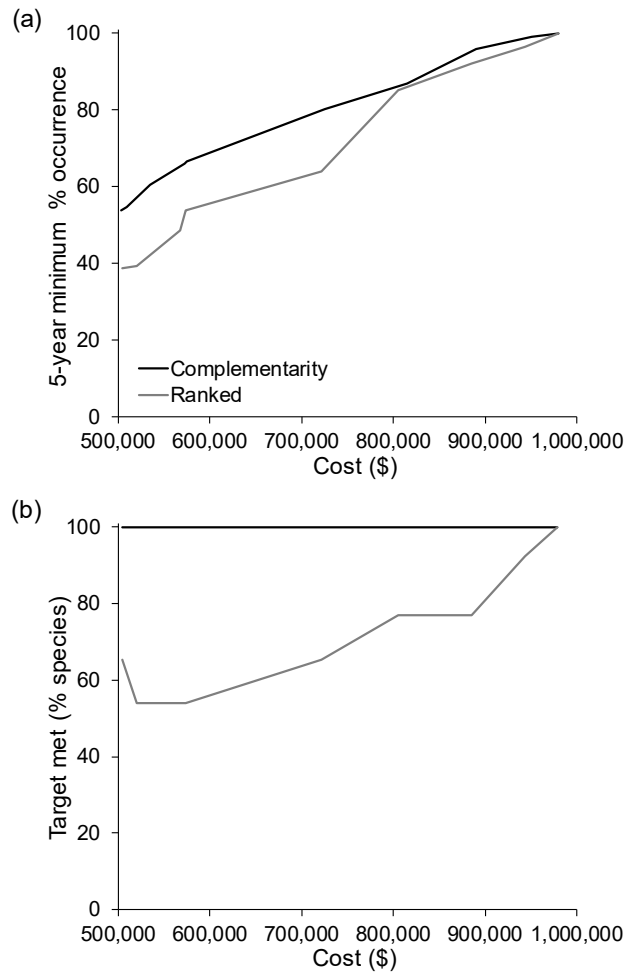
Figure 3: Comparison of dynamic complementarity and ranked approaches to maximise landscape-scale occurrence of species of conservation concern for (a) 5-year mean minimum percent occurrence, and (b) targets met. Each line represents 10% to 100% representation targets.
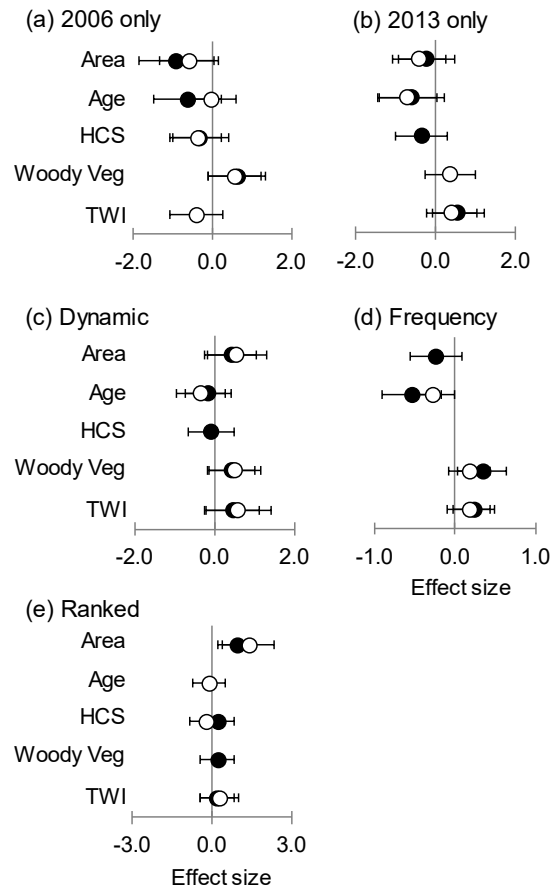
Figure 4: Summary of model-averaged effect sizes (and 95% confidence intervals) for terms in the top-ranked models (ΔAICc≤ 2) for 30% (closed circles) and 60% (open circles) representation targets. See Table 2 for a description of model terms. See Figure S1 for plots for 2008 only, 2009 only and 2011 only models.

Supporting Information

Table S1: Summary of explanatory variables considered in the analyses, and example of a previous study that has found the variable to be important in explaining bird diversity in restoration plantings.

Variable	Definition	Mean (Range)	Example studies
Cost	Establishment cost	$16, 052 ($4, 948 – $75,869)	Polyakov et al. 2015
Age	Number of years since the establishment of the planting (since 2006).	11 (0 – 44)	Lindenmayer et al. 2010 Munro et al. 2011
Area	Size of planting (ha).	4.24 (0.3 – 60.3)	Kavanagh et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011
Width	Width of planting (m).	65.16 (10 – 300)	Kinross 2004 Lindenmayer et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011
Habitat complexity score (HCS)	Vegetation structural complexity was based on vegetation data collected in 2007/08 and 2013: (i) the percent cover of overstorey, midstorey and understorey vegetation, the number of logs per ha, and the presence of large trees (> 50 cm diameter at breast height) were recorded within three 20 x 20 m plots located at the 0 m, 100 m and 200 m transect points; and (ii) the percent cover of native grass, exotic grass, exotic perennials, broadleaf weeds, forbs, leaf litter, and moss and lichen were recorded within twelve 1 m x 1 m quadrats located at the corners of the plots. A combined site-level habitat complexity score was calculated from these data, following Munro et al. (2011) (Table S2).	18 (9 – 29)	Lindenmayer et al. 2010 Munro et al. 2011
Woody vegetation (WoodyVeg)	Percentage of vegetation cover within a 1 km buffer from the 100 m transect point. Derived	5.45% (0.00% – 23.00%)	Kavanagh et al. 2007 Lindenmayer et al. 2010 Munro et al. 2011

Variable	Definition	Mean (Range)	Example studies
	from Landsat satellite imagery (Danaher 2011).		Radford et al. 2005
Topographic wetness index (TWI)	Position in landscape, ranging from ridge tops to valley floors. Derived from a 20 m resolution Digital Elevation Model (DEM) (Montague-Drake et al. 2011), and calculated at the 100 m transect point	0.61 (-2.68 – 10.23)	Lindenmayer et al. 2010 Montague-Drake et al. 2011

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Table S2: Habitat complexity score (HCS). Planting HCS was the sum of the scores for each element.

Score	Strata % cover*	Logs/ha	Trees > 50 cm/ha
0	< 1%	< 1	< 1
1	1-5%	1-10	
2	6-30%	11-50	
3	31-70%	51-100	
4	> 70%	> 100	≥ 1

*Strata includes overstorey, midstorey, understorey and ground layer (native tussock, exotic tussock, exotic grass, broadleaf weeds, forbs, and leaf litter).

Table S3: Costs of materials and labour for fencing and direct-seeding of restoration sites

Item	Description	Rate ($AUD)
Fencing	Fencing materials and labour	$10,000/km
Direct-seeding - materials	Seed, machinery	< 2 ha = $750/ha
		2-4 ha = $625/ha
		>4 ha = $550/ha
Direct-seeding - labour	Labour, site preparation	$77.68/ha

Table S4: Woodland bird species of conservation concern, justification for inclusion and number of observations between 2006 and 2013. 'Legislation': listed as threatened in NSW under the *Threatened Species Conservation Act 1995* (this also captures relevant nationally-listed threatened species) and/or 'Atlas': identified as having a >20% decrease in South West Slopes bioregion reporting rate between the first and second Atlas of Australian Birds.

Common name	Scientific name	Source	Records
Black-chinned Honeyeater	Melithreptus gularis	Legislation	6
Brown Songlark	*Cincloramphus cruralis*	Atlas	56
Brown Treecreeper	*Climacteris picumnus*	Legislation	8
Cockatiel	*Nymphicus hollandicus*	Atlas	15
Crested Shrike-tit	*Falcunculus frontatus*	Atlas	30
Diamond Firetail	*Stagonopleura guttata*	Legislation	21
Dollarbird	*Eurystomus orientalis*	Atlas	2
Dusky Woodswallow	*Artamus cyanopterus*	Atlas	10
Fairy Martin	*Petrochelidon ariel*	Atlas	5
Grey-crowned Babbler	*Pomatostomus temporalis*	Legislation	11
Jacky Winter	*Microeca fascinans*	Atlas	3
Little Lorikeet	*Glossopsitta pusilla*	Legislation	3
Masked Woodswallow	*Artamus personatus*	Atlas	7
Pied Butcherbird	*Cracticus nigrogularis*	Atlas	5
Rainbow Bee-eater	*Merops ornatus*	Atlas	13
Restless Flycatcher	*Myiagra inquieta*	Atlas	9
Scarlet Robin	*Petroica boodang*	Legislation	2
Southern Whiteface	*Aphelocephala leucopsis*	Atlas	10
Speckled Warbler	*Chthonicola sagittata*	Legislation	9
Superb Parrot	*Polytelis swainsonii*	Legislation	19
Weebill	*Smicrornis brevirostris*	Atlas	66
White-browed Woodswallow	*Artamus superciliosus*	Atlas	54
White-fronted Chat	*Epthianura albifrons*	Legislation	8
White-winged Triller	*Lalage sueurii*	Atlas	46
Yellow-rumped Thornbill	*Acanthiza chrysorrhoa*	Atlas	119
Zebra Finch	*Taeniopygia guttata*	Atlas	2

Table S5 Comparison of scenarios with cost included and excluded, for the 30% and 60% species occurrence targets. The locations of plantings selected under the two scenarios were similar (Bray-Curtis dissimilarity 13% and 23% for the 30% target and 60% target, respectively).

Scenario	Cost	Plantings	Area (ha)	% Occurrence	Target met
30%: cost included	$535,125.80	32	185.00	60.63	100
30%: cost excluded	$591,778.60	30	203.80	60.28	100
60%: cost included	$725,628.00	43	222.20	80.19	100
60%: cost excluded	$754,132.20	42	227.50	80.53	100

Table S6. Mean (SD) selection frequencies of plantings selected in the best solutions for each representation target and those not selected.

	Best solution	
Target	Selected	Not selected
10%	98.71 (6.42)	1.06 (5.92)
20%	96.34 (13.32)	3.22 (9.90)
30%	96.88 (10.93)	3.55 (10.92)
40%	98.00 (7.91)	3.07 (19.93)
50%	98.06 (7.80)	3.19 (9.03)
60%	98.21 (7.56)	4.28 (11.40)
70%	96.25 (12.42)	13.92 (17.29)
80%	97.22 (11.57)	24.00 (21.76)
90%	99.62 (2.89)	19.67 (15.31)
100%	100.00 (0.00)	-

Table S7. Summary of dynamic and static complementarity scenarios

Target (%)	Dynamic			2006			2008			2009			2011			2013		
	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)
10	$503,891	54	100	$114,068	5	15	$209,546	11	31	$156,488	4	19	$281,986	11	35	$166,364	4	12
20	$509,593	55	100	$123,890	5	12	$247,124	17	42	$166,002	5	4	$297,123	11	31	$177,061	6	19
30	$535,126	61	100	$163,907	7	12	$283,631	19	38	$216,123	8	8	$332,201	17	27	$212,421	11	15
40	$573,122	66	100	$195,243	8	4	$330,638	33	50	$266,335	9	4	$370,808	19	27	$257,570	12	4
50	$575,591	66	100	$229,085	13	12	$381,571	34	42	$305,878	10	4	$395,025	20	23	$300,573	22	15
60	$725,628	80	100	$332,518	13	8	$549,920	41	31	$448,961	21	4	$480,195	35	23	$395,403	28	8
70	$814,979	87	100	$397,461	25	8	$620,428	48	27	$534,510	26	4	$525,713	36	15	$459,828	36	12
80	$889,818	96	100	$503,252	31	8	$690,304	57	27	$606,834	28	4	$661,279	54	27	$572,794	45	15
90	$951,035	99	100	$574,287	38	8	$789,620	76	35	$704,481	37	8	$746,849	59	23	$751,666	57	12
100	$979,198	100	100	$633,418	51	19	$890,734	89	62	$816,398	64	31	$788,795	61	23	$870,880	71	27

Table S8. Summary of dynamic complementarity and ranked scenarios.

| Target (%)	Dynamic			Species-richness ranked			Species-richness / cost ranked		
	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)	Cost ($AUD)	5-yr. min occ. (%)	Target met (%)
10	$503,891	54	100	$504,144	39	65	$502,903	51	77
20	$509,593	55	100	$504,144	39	65	$502,903	51	77
30	$535,126	61	100	$520,595	39	54	$525,424	51	73
40	$573,122	66	100	$567,577	49	54	$565,378	55	81
50	$575,591	66	100	$574,061	54	54	$565,378	55	81
60	$725,628	80	100	$721,545	64	65	$701,886	73	69
70	$814,979	87	100	$805,461	85	77	$819,627	86	81
80	$889,818	96	100	$885,168	92	77	$865,478	91	88
90	$951,035	99	100	$943,044	96	92	$892,227	91	85
100	$979,198	100	100	$979,198	100	100	$979,198	100	100
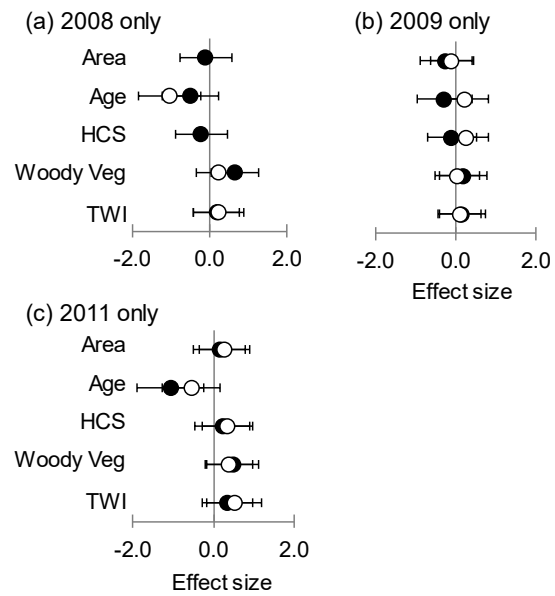
Figure S1. Summary of model-averaged effect sizes (and 95% confidence intervals) for terms in the top-ranked models (ΔAICc≤ 2) for 30% (closed circles) and 60% (open circles) representation targets. 'Area' is planting area, 'Age' is years since planting establishment, 'HCS' (habitat complexity score) represents vegetation structural complexity, 'Woody Veg' is percentage of vegetation cover within 1000 m, and 'TWI' (topographic wetness index) represents position in landscape ranging from ridges to valley floors. See Table S1 for a full description of model terms.