1 **Contrasting effects of pine plantations on two skinks: results from a large scale 'natural**

2 **experiment' in Australia**

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12   **Abstract**

13   The aim of our study was to quantify the long-term response of two lizard species to the

14   transformation of the matrix surrounding remnant habitat patches (from agricultural land to

15   pine plantations).

16   We used a large scale (115 sites), long-term (16 years) fully replicated and controlled

17   landscape scale 'natural experiment' (the 'Nanangroe experiment', Australia) to compare

18   the response of lizard populations to plantation establishment. The study entailed detailed

19   surveys of reptiles in 50 eucalypt patches surrounded by maturing Pine (*Pinus radiata*)

20   plantations (*treatments*) and populations inhabiting 55 eucalypt patches surrounded by

21   grazing land (*controls*).

22   We found that the Eastern Three-toed Earless Skink (*Hemiergis talbingoensis*) was

23   advantaged by the establishment of the pine plantations (increasing colonization of eucalypt

24   patches embedded within plantations). In contrast, we found that the Southern Rainbow

25   Skink (*Carlia tetradactyla*) was negatively affected by surrounding areas of maturing

26   plantations. Thus our results show that plantations acted as a barrier for one species and

27   increased connectivity for the other. We suggest that leaving areas of land devoid of pines

28   between remnant eucalypt patches may enhance the connectivity for species that respond

29   negatively to plantation establishment, while maintaining the beneficial increase in

30   connectivity for others.

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Introduction

The conversion of agricultural land to forest plantations is a major driver of global land use change (Foley *et al.*, 2005). To date, planted forest (including production-oriented plantations and other types of planted forest (FAO, 2010) are present in most vegetated countries. Planted forests cover approximately 264 billion hectares, 7% of global forest cover, roughly equivalent to half the Amazonian rainforest. Recent decades have witnessed substantial increases in planted forest (e.g. a global increase of 5 million hectares/year was recorded by FAO between 2000-2010) due to an increased demand for wood and carbon storage (Jackson *et al.*, 2005; Paquette & Messier, 2010). Considering that large portions of the planet are covered by forest plantations, understanding their impact on biodiversity is a research priority (Brockerhoff *et al.*, 2008; Felton *et al.*, 2010).

Previous research on biodiversity on plantations focused mainly on birds (Renjifo, 2001; Mortelliti, Westgate & Lindenmayer, 2014) whereas reptiles have attracted considerably less attention (Gardner *et al.*, 2007). Previous studies on reptiles have found that plantations tend to decrease species richness and diversity (Glor *et al.*, 2001; Loehle *et al.*, 2005; Amo, López & Martín, 2006; Gardner *et al.*, 2007; Lindenmayer *et al.*, 2008) but see (Vonesh, 2001). The magnitude of effects is context-specific and varies with the type of plantations (e.g. eucalypt vs pine), the habitat of target species, and microhabitat availability within the plantation (Glor *et al.*, 2001; Vonesh, 2001; Loehle *et al.*, 2005; Kanowski *et al.*, 2006). Previous studies have been short-term and observational and have focused mainly on species occurrence within the plantations (e.g. by contrasting assemblages in natural vs planted forests). Few studies have focused on the effects of plantations as 'landscape context' (i.e. on reptiles inhabiting patches of remnant native vegetation embedded within a plantation). An additional knowledge gap is that, to date, no study within plantations has explicitly focused on colonization/extinction parameters, which are key ecological variables in fragmentation studies (Hanski & Gaggiotti, 2004). Focusing on population turnover allows a more in-depth understanding of population dynamics than the "static" occupancy studies which are based on 'snapshot' presence/absence data and also allows for a separate evaluation of what is affecting population establishment (e.g. colonization) and what is affecting persistence (e.g. the extinction risk).

65 To address key knowledge gaps about reptile responses to plantation establishment we

66 used a large-scale (115 sites over 30000 ha), long-term (16 year) fully replicated and

67 controlled landscape experiment, conducted in south-eastern Australia (the 'Nanangroe

68 experiment'). The aim of the 'Nanangroe experiment ' (Lindenmayer, 2009) is to compare

69 responses of target species inhabiting 50 *treatment* eucalypt patches surrounded by

70 maturing Pine (*Pinus radiata*) plantations, with populations inhabiting 55 *control* eucalypt

71 patches surrounded by grazing land (Fig. 1). A unique feature of the Nanangroe study is that

72 both treatments and controls have remained unmodified throughout the study period,

73 being the first study of its kind to focus on the effects of the 'outside' matrix on target

74 populations within eucalypt patches.

75 We chose two skinks (Scincidae) as target species: the Southern Rainbow Skink *Carlia*

76 *tetradactyla* and the Eastern Three-toed Earless Skink *Hemiergis talbingoensis*. The aim of

77 our study was to quantify the long-term response of these two target species to the

78 transformation of the matrix surrounding remnant habitat patches (from agricultural land to

79 pine plantations).

80

81 **Materials and methods**

82 *Study area*

83 Our research was conducted in the Nanangroe area (New South Wales, South-eastern

84 Australia; Fig. 1). The Nanangroe area lies approximately 70 km north-west of Canberra (co-

85 ordinates 34°54' - 35°4' and 148°32' - 148°18' E, altitudinal range: 250-750 m a.s.l.), covers

86 approximately 30 km² and is characterised by hot summers and cool winters (temperate

87 climate). The native vegetation (more than 80% of the original temperate eucalypt

88 woodland has been cleared for grazing) is characterised by open woodlands dominated by

89 red box (*Eucalyptus polyanthemos*), white box (*E. albens*), yellow box (*E. melliodora*),

90 Blakely's red gum (*E. blakleyi*) and red stringybark (*E. macrorhyncha*).

91

92 *Experimental design of the Nanangroe project*

93 Pine plantations (*Pinus radiata*) began being established in Nanangroe in late 1998 by Forest

94 NSW for the production of timber and pulp (paper) and for carbon sequestration purposes.

95 Approximately 70 eucalypt patches of variable sizes were retained during the establishment

96   of the plantation, which were progressively surrounded by the emerging artificial landscape

97   (Fig 1).

98   The selection of sites followed a replicated, random stratified procedure. The stratification

99   was based on three criteria : a) size of the eucalypt habitat patches; b) age-cohort of the

100   surrounding pine plantation; c) number of boundaries between patches and surrounding

101   pine stands.

102   The study design (Fig 1, Table S2) included: a) 50 woodland *treatment* sites (sites within

103   eucalypt patches surrounded by *Pinus radiata* plantation); b) 55 woodland *control* sites

104   (sites within eucalypt patches surrounded by grazing land); c) 10 pine control sites (sites

105   within the plantation).

106   For each of the treatment and control sites, we selected a series of replicates based on the

107   size of the habitat patch. Patches were grouped according to patch size intervals: 0.5-0.9 ha

108   (13 replicates), 1.0-2.4 ha (19 replicates), 2.5-4.9 ha (15 replicates), >5.0 ha (3 replicates).

109   Eucalypt patches were surrounded by pines belonging to two age-cohorts (cohort 1= pines

110   planted in 1998; cohort 2= pines planted in 2000). Of the 50 woodland treatments, nine had

111   1-2 open boundaries with grazed land, whereas the remaining patches were completely

112   surrounded by pines. A summary of the experimental design is provided in Table S2. The

113   woodland treatment sites were matched with 55 control sites surrounded by grazing land

114   (Table S2). Domestic livestock grazing (sheep and cows) continued in all sites throughout

115   the whole study period, thereby eliminating potential confounding effects between

116   treatments.

117

118   *Reptile surveys*

119   We used arrays of artificial refuges (Michael *et al.*, 2012) to survey the distribution

120   (presence/absence) of the two target lizard species in each of the 115 sites. Artificial refuges

121   (ARs) were composed of: (1) one double-layered stack of corrugated galvanised steel, (2)

122   four 1-m long *E. dalrympleana* fence posts covered in mesh, and (3) four concrete roof tiles

123   (32 cm*42 cm). Over time, the fence posts decayed, so in 2010 we installed four 1.2 m long

124   railway sleepers at each site (Fig. 2).

125   At each site, we established two reptile monitoring stations (AR arrays) located 100 m apart.

126   Surveys were conducted in 1999, 2000, 2003, 2006, 2011 and 2013. Surveys were

127   completed during spring (August) each survey year and were conducted on clear sunny days

128 between 0900 hrs and 1400 hrs by the same group of experienced herpetologists from The

129 Australian National University.

130 Thirteen reptile species were captured during the study, of which *C. tetradactyla* and *H.*

131 *talbingoensis* were the two most abundant species. Capture rates for other species was low

132 and did not allow us to fit occupancy models (see below).

133

134 *Data analysis*

135 We selected a restricted set of predictor variables to be included in the data analyses (see

136 'occupancy models' below) in addition to "treatment" and "time" effect. These included:

137 vegetation type and eucalypt tree cover surrounding the site. Vegetation type categories

138 were measured in the field and were based on dominant or co-dominant tree species (Table

139 S3). Eucalypt tree cover was measured in a circular buffer (250 m radius) centred in each

140 site. Tree cover included habitat patches and isolated trees and was measured by using

141 digitised aerial photography in ArcGIS 10.1.

142 We chose to use the tree cover in a buffer rather than patch size because scattered trees

143 play a crucial role in determining distribution patterns of Australian native fauna (Fischer,

144 Stott & Law, 2010). Furthermore, buffer-based measurements of habitat amount have been

145 recommended as single predictor variables in fragmentation studies (instead of patch size

146 and isolation) (Fahrig, 2013).

147

148 <u>Occupancy models</u>

149 False absences (a species was present in a site but not detected) are a key source of bias in

150 distribution studies, particularly for studies focusing on terrestrial vertebrate species

151 (MacKenzie, 2005). To control for possible false absences, we fitted multiple season

152 occupancy models to presence/absence data from our reptile surveys thus allowing the

153 estimation of colonisation/extinction probabilities (MacKenzie, Nichols & Hines, 2003). We

154 focused our occupancy analyses on the 105 woodland eucalypt patches (50 woodland

155 treatments and 55 woodland controls). We retained 10 pine control sites for separate

156 analyses (detailed below).

157 We defined each eucalypt patch as a *site* (*sensu* (MacKenzie *et al.*, 2003)). We considered a

158 *visit* to a site the inspection of one of the two AR arrays. Therefore, each site was surveyed

159 twice on the same day during the same year. Following (MacKenzie *et al.*, 2003),

160 populations were assumed to be closed within each survey year (i.e. between visits) and

161 open to colonization/extinction between survey years.

162 We adopted the following protocol for fitting models:

163   1) The detection probability (p) parameter was either held constant across years or

164   modelled as a function of year. The variable "year" was included in two different

165   ways: a) we included it as a categorical variable to account for possible unmeasured

166   year-specific factors affecting the detectability of our two target species; b) we

167   included it as a continuous variable to check for possible trends (i.e. an increase or

168   decrease) in populations. The relative best model was selected through the Akaike

169   Information Criterion (AIC (Burnham & Anderson, 2002). The variable included in the

170   top ranking detection model was retained in all the following steps:

171   2) In the second step of our protocol, we selected the most important variables

172   influencing Ψ_1 (probability of a site being occupied in the first study year). Ψ_1 was

173   modelled as a function of vegetation type and tree cover in the 250 m buffer). The

174   variable included in the top ranking occupancy model was retained in the following

175   step.

176   3) We modelled the *probability of colonisation* (Υ) and the *probability of extinction* (ε)

177   as functions of the following variables: treatment (a site surrounded by pines vs

178   control), year since the beginning of the study (continuous variable) and tree cover

179   in the 250 m buffer. We also included two-way interactions for these variables (e.g.

180   treatment* year) and included models with different effects for colonisation and

181   extinctions: e.g. Υ(treatment*time), ε(tree cover).

182

183 The variable "year" was measured as time since the beginning of the study, which in the

184 case of treatments also acted as a proxy for the time since plantation establishment. We

185 were unable to identify a method to include a time covariate for treatments only (the value

186 zero in the control sites would not be meaningful). Similarly, distinguishing between the two

187 cohorts of plantation establishment and simultaneously modelling control sites was not

188 possible. To avoid conducting separate analyses for treatments and controls, we opted to

189 use the variable time since start of the study, and acknowledge that this was only a proxy

190 for time since the establishment of the plantation. Occupancy models were fitted using the

191 *unmarked* package for R (Fiske & Chandler, 2011).

192   We based our inference on model averaged estimates (including models within 2 ΔAIC

193   (Burnham & Anderson, 2002). Goodness of fit of each model was measured using

194   Nagelkerke's R^2.

195

196   Our experiment was designed to limit the risk of spatial dependence between sites (e.g.

197   spatial autocorrelation and pseudo-replication): a) four independent plantations were

198   included as treatments; b) woodland control sites were distributed amongst six different

199   farms; c) average distance between sites was 9.3 km (range 0.11-21.7 km). Notwithstanding,

200   we checked for spatial autocorrelation in the residuals of the most parameterised model by

201   using a spline correlogram (Zuur *et al.*, 2009). We used the R package ncf to produce

202   correlograms (Bjørnstad, 2009). We calculated correlograms for each of the six time periods

203   and could not find evidence of spatial autocorrelation for either of the two target species.

204

205   *Reptile use of the matrix sites*.

206   To evaluate the relative capabilities of each species to use/cross the pine matrix, we

207   calculated an index of the relative abundance of each skink in the pine matrix (i.e. the 10

208   sampling sites located within the pine matrix) by summing the number of detections for

209   each species in these sites. We tested whether the relative abundance of the two target

210   species in the matrix were significantly different through a Kruskal-Wallis test. Effect size

211   was calculated through a post-hoc Mann-Whitney test (i.e. it is not possible to estimate

212   effect size for the Kruskal-Wallis test). Both these analyses we conducted using the mass

213   package in R (R Core Team, 2013).

214

215   **Results**

216   Over the 16-year duration of our study, we detected *H. talbingoensis* 253 times in the

217   eucalypt patches (151 detections in treatments and 102 in controls), whereas we detected

218   *C. tetradactyla* on 117 occasions in eucalypt patches (39 detections in treatments and 78 in

219   controls).

220   *Occupancy models.* The probability of detecting *C. tetradactyla* varied each year (i.e. year as

221   categorical variable, Fig S3) whereas the probability of detecting *H. talbingoensis* increased

222   over time (i.e. year as continuous variable, Fig S3). The probability of a site being occupied in

8

223   the first sampling season was dependent on vegetation type for both species (Table 1). The

224   variable 'treatment' was included as predictor of colonization probability in the top ranked

225   models for both species. However, the two lizard species showed strongly contrasting

226   responses to the establishment of the pine plantations surrounding the eucalypt patches.

227   For *C. tetradactyla,* colonization probability was lower in treatments, whereas in *H.*

228   *talbingoensis* the probability of colonization was higher in treatments (Fig 3; Table 1). In the

229   top ranked model for both species, the extinction probability was constant (Table 1).

230   Predictors for probability of extinction included in the top model set (e.g. within ΔAIC<2 but

231   with less support compared to the first model) included year (positive beta = increase with

232   time) and habitat (negative beta = more habitat within the buffer decreased extinction risk)

233   for *C. tetradactyla* and treatment (negative beta = treatment sites had lower extinction risk),

234   year (positive beta) and habitat (negative beta) for *H. talbingoensis*.

235   *Captures in the matrix.* Throughout the study period, we captured a higher number of *H.*

236   *talbingoensis* (seven captures) than *C. tetradactyla* (one capture) in the pine matrix.

237   However, this difference was not statistically significant (Kruskall-Wallis χ^2 = 2.53, df = 1, *p* =

238   0.1; medium effect size: r = 0.35).

239

240   **Discussion**

241   Plantation forestry is widely recognised as major driver of global change (Foley *et al.*, 2005).

242   We used a large-scale and long-term landscape transformation experiment to identify the

243   contrasting response of two lizard species to the broad-scale establishment of pine

244   plantations. A unique aspect of our study was that we focused on the effects of changes

245   occurring in the matrix on lizard populations inhabiting eucalypt patches. We found that

246   pine plantations increased connectivity (i.e. colonization probability increased in the

247   eucalypt patches surrounded by plantation) for one species (*H. talbingoensis*), but acted as a

248   barrier (i.e. the probability of colonization in eucalypt patches was lower in treatment sites)

249   for the other species (*C. tetradactyla*).

250

251   *Biological interpretation of the models*

252   The small taxonomic scope of our research (2 species) limits our inference on the possible

253   mechanisms that may have driven the observed patterns (Garland & Adolph, 1994).

254   Below we list three (non mutually exclusive) key hypotheses that may explain the

255   contrasting pattern that we have observed (but see also Table S1 for additional differences

256   between the target species). These hypotheses should be field-tested by future studies with

257   a broader taxonomic scope.

258   *Hypothesis 1*). *Thermoregulation.* The Southern Rainbow Skink (*C. tetradactyla)* is a

259   heliothermic species (i.e. it gains heat by short wavelength solar radiation) whereas the

260   Eastern Three-toed Earless Skink (*Hemiergis talbingoensis*) is a thigmotherm (it gains heat by

261   direct contact with a warm substratum) (Garrick, 2008) (Table S1).

262   (Fischer, Lindenmayer & Cowling, 2003a) found that *C. tetradactyla* prefers open woodlands

263   to woodland with more closed canopy, perhaps due to its heliothermic requirements, which

264   may suggest that high levels of shade associated with pine plantations (Porté, Huard &

265   Dreyfus, 2004; Mott, Alford & Schwarzkopf, 2010) act as a barrier for this species and cause

266   decreased colonization rates in the treatments compared with controls.

267   Although *H. talbingoensis* also inhabits eucalypt woodland (Fischer *et al.*, 2005a; Kay *et al.*,

268   2013), compared to *C. tetradactyla,* it is better adapted to living in cold and humid

269   environments. The field active body temperature of *H. talbingoensis* is 20.3 C° (range =

270   17.3 C° - 23 C°), which is considerably lower than heliothermic skink species (Greer, 1989).

271   Importantly, the critical thermal minimum and maximum temperature range (CTmin = 6.8

272   C°, CTmax = 38.6 C°) is particularly low (data from (Bennett & John-Alder, 1986). In

273   accordance with these findings, (Michael, Cunningham & Lindenmayer, 2011) found the

274   predicted response of *H. talbingoensis* to eucalypt regrowth (i.e. dense vegetation cover) to

275   be positive, presumably because increased shade levels associated with regenerating

276   vegetation created conditions consistent with the species autecology. Thus, according to

277   this hypothesis, mode of thermoregulation (heliothermy vs thigmothermy) may be a key

278   driver of their response to pine plantations (Mott *et al.*, 2010). We emphasise that to test

279   this hypothesis detailed measurements of the thermal environments should be conducted

280   (Mott *et al.*, 2010).

281   *Hypothesis 2) Reproductive mode.* Another key difference between the two target species

282   that may help to explain the strongly contrasting responses observed is the mode of

283   reproduction (Table S1). *Carlia tetradactyla* is oviparous (thus the pine matrix may be too

284   cold for incubation) whereas *H. talbingoensis* is viviparous (i.e. gives birth to live young, thus

285   may potentially breed in the pine matrix; Table S1). Previous studies have shown that

286 availability of warm sites is critical for oviparous skinks (Shine, Elphick & Harlow, 1997) thus

287 reproductive mode may have played an important role in determining the observed

288 patterns (but see later discussion on matrix use).

289 *Hypothesis 3) Prey availability and predation.* Differential prey availability or predatory risk

290 caused by the plantations may also explain the differences observed. Pine plantation may

291 host different invertebrates (Barbaro *et al.*, 2012), which in turn may favour matrix-crossing

292 for one species compared to the other. Similarly, the low understory of pine plantations

293 may increase predation risk for *C. tetradactyla* but not for the burrowing skink (*H.*

294 *talbingoensis*) and thus create a barrier effect for one species only.

295 A critical aspect that needs further clarification is the reptile use of the matrix sites.

296 Clarifying this issue will help identifying the mechanisms determining the contrasting

297 responses observed. We found captures of *H. talbingoensis* within the plantations were

298 higher than captures of *C. tetradactyla*, suggesting that this species has a higher capability

299 of moving through or persisting in pine plantations. The low p value (p = 0.1) but medium

300 effect size (0.35) suggest that lack of statistical significance may be due to poor statistical

301 power caused by low capture rates. Further analyses are therefore warranted to reach

302 definitive conclusions. The extremely low number of captures suggests that *H. talbingoensis*

303 occasionally moves through the plantation matrix (rather than living within them at low

304 population density; see also (Fischer *et al.*, 2005b) for similar conclusions). Several factors

305 could determine different matrix crossing capabilities, such as: a) thermoregulation, b)

306 tolerance to dry conditions (i.e. higher soil moisture in plantations may facilitate movement

307 of the fossorial species c) different and species-specific perceptual ranges in the two types

308 of matrix (Sozio, Mortelliti & Boitani, 2013). Nevertheless, we emphasise that further

309 empirical evidence is required to reach more definitive conclusions on this issue.

310 Our occupancy analyses revealed that plantations affected colonization rather than

311 extinction processes. However, we found some evidence for lower extinction risk of *H.*

312 *talbingoensis* in treatment sites (Table 1). It is likely that a longer study (e.g. >16 years) may

313 reveal an important influence of plantation establishment on local extinction risk, as a

314 decrease in colonization will likely lead to lack of immigration in already occupied patches

315 (and thus to an increase in local extinction risk)(Hanski & Gaggiotti, 2004). The patterns we

316 have discovered were not detected in previous studies in this area (Lindenmayer *et al.*,

317 2008) possibly because the present study was substantially longer (16 years vs 8 years).

318   Furthermore, our study was focused on different ecological variables (population turnover

319   rather than occupancy).

320

321   *Implications for conservation*

322   Our study is unique in focusing on the effects that plantations have on populations

323   inhabiting eucalypt patches surrounded by plantations. Consequently, our results are

324   applicable to plantations containing remnant patches. We emphasise that pines are the

325   most widespread species in plantations worldwide (20% of plantations) (Brockerhoff *et al.*,

326   2008) thus making our study system highly representative.

327   We foresee a major implication of our study for reptile conservation within transformed

328   landscapes: pine plantations may act as a barrier (decreasing colonisation probability) for

329   some species (*C. tetradactyla* in our study) and increase connectivity for others (*H.*

330   *talbingoensis* in our study). We suggest the establishment of habitat corridors – in this case

331   open woodland or open canopy areas – may help mitigate the negative impact of

332   plantations on these species (see also (Pryke & Samways, 2001; Bertoncelj & Dolman, 2012,

333   2013) for similar conclusions on other taxa). The colonization rate of control patches

334   (surrounded by grazing) was higher than treatment sites, confirming that *C. tetradactyla* is

335   relatively good at crossing a matrix of open fields (Fischer *et al.*, 2005a) especially when not

336   heavily grazed (Fischer, Lindenmayer & Cowling, 2003b). We therefore suggest that leaving

337   areas without pines between remnant patches may enhance connectivity for this species.

338   Such a conservation action may be substantially cheaper than creating revegetated

339   corridors, only resulting in foregone profit, rather than both missed profit and costs of

340   habitat restoration. Improving the quality of microhabitats (e.g. providing debris, rocks etc)

341   also may improve the suitability of plantations for reptile species (Amo *et al.*, 2006;

342   Kanowski *et al.*, 2006).

343   As our study was conducted on only two species we cannot provide clear explanations on

344   which mechanisms (e.g. which life history traits) determined the contrasting responses

345   observed. We hypothesise that mode of thermoregulation, reproductive mode and prey

346   availability or predation risk may be potential drivers. The possibility of predicting the

347   response of species to land use change based on life-history traits is a major challenge in

348   conservation biology (Henle *et al.*, 2004; Koh, Sodhi & Brook, 2004; Ockinger *et al.*, 2010).

349   Nevertheless further multi-species studies are needed to test our hypotheses and thus to

350   generalize our findings.

351   The results of our broad-scale landscape transformation experiment show how landscape

352   context surrounding remnant habitat patches affects the long-term population dynamics of

353   lizard populations. Converting agricultural land to pine plantations may favour some species

354   and increases the colonization rate of previously unoccupied eucalypt patches but may also

355   create a barrier for other species. We suggest that leaving land devoid of pines between

356   remnant patches may enhance the connectivity for some species while maintaining the

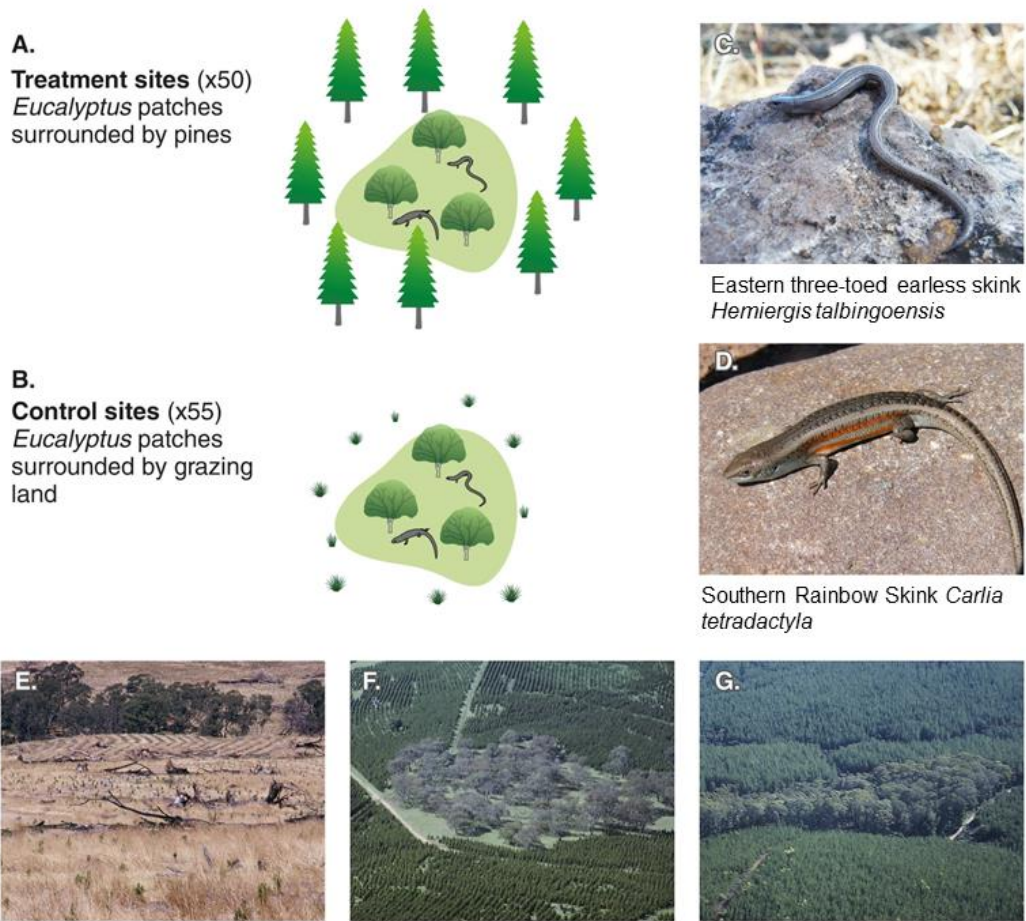357   beneficial increase in connectivity for others.

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480

481

A.
Treatment sites (x50)
Eucalyptus patches
surrounded by pines

B.
Control sites (x55)
Eucalyptus patches
surrounded by grazing
land

C.
Eastern three-toed earless skink
Hemiergis talbingoensis

D.
Southern Rainbow Skink *Carlia tetradactyla*

482

Fig. 1. Features of the 'Nanangroe experiment' (New South Wales, Australia). A) Treatment sites are eucalypt habitat patches surrounded by pine plantations. B) Control sites are eucalypt habitat patches surrounded by grazing land; C-D) Target species; E) Eucalypt habitat patch surrounded by grazing land (small growing pines visible); F-G) Later stages of *Pinus radiata* plantation development; treatments progressively surrounded by forested landscape.
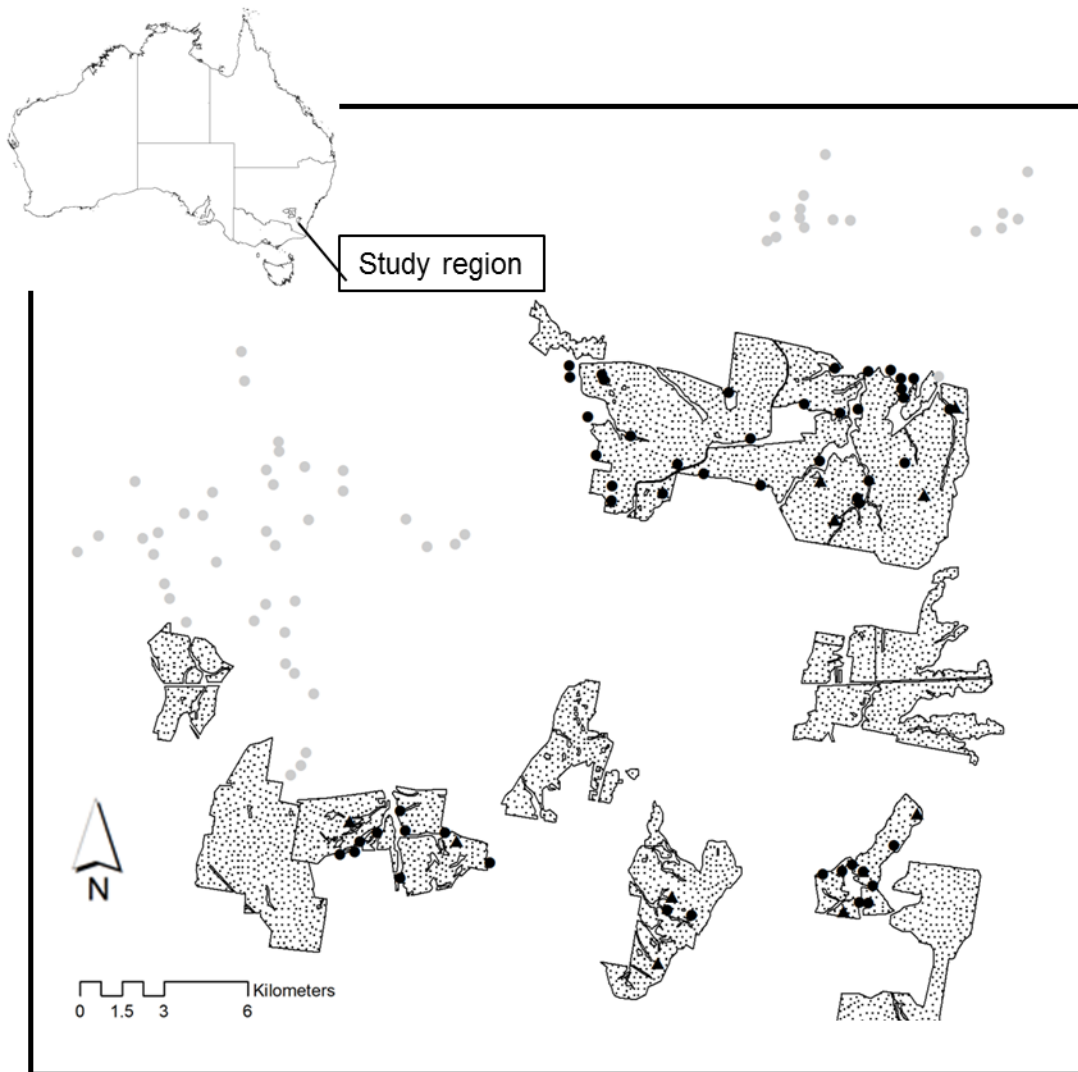
Fig. 2. Map of the study area (Nanangroe, New South Wales, Australia) with sampling sites (grey circles = controls; black circles = treatments, black triangles= pine). The dotted area corresponds to pine plantations, whereas the white area corresponds to grazed field.

Carlia tetradactyla   Hemiergis talbingoensis
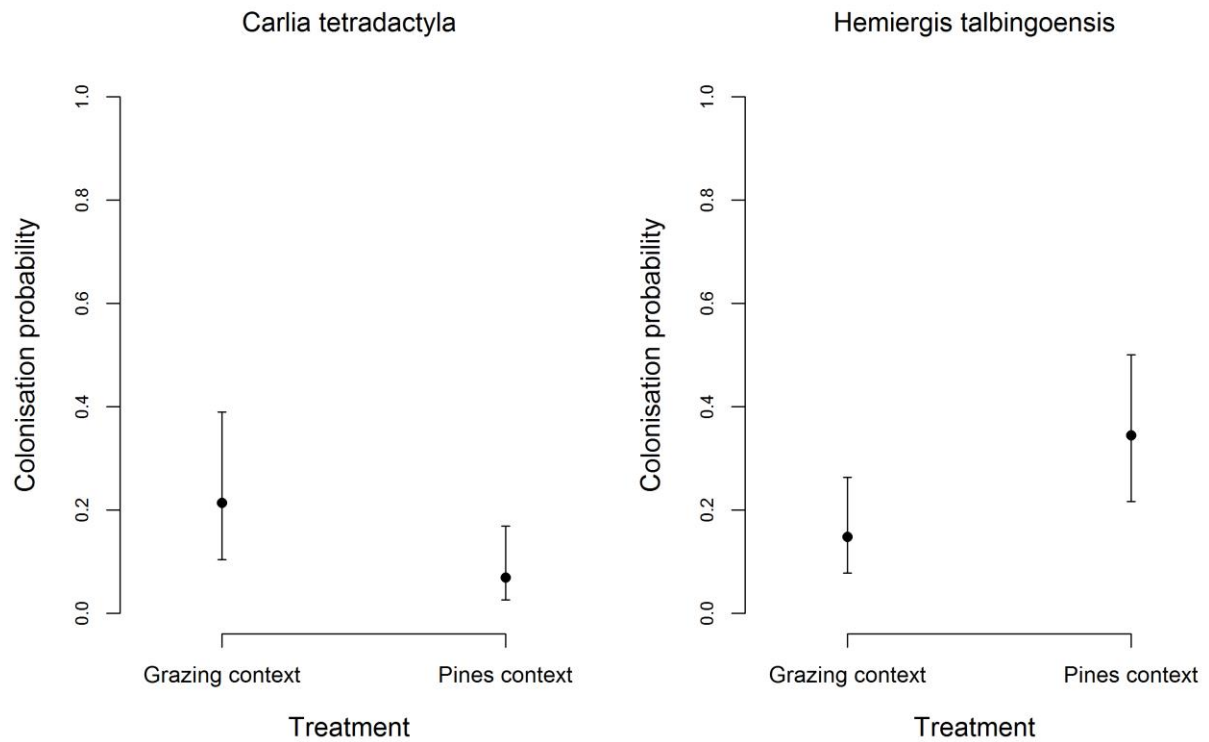
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495   Fig. 3. Model predictions (including 95% confidence intervals) based on model averaged

496   estimates of top ranking models (ΔAIC<2) for the Southern Rainbow Skink *Carlia*

497   *tetradactyla* and the Eastern Three-toed Earless Skink *Hemiergis talbingoensis* . *Carlia*

498   *tetradactyla* responded negatively to pine plantations (decrease in colonisation probability

499   in sites surrounded by pine plantations), whereas *H. talbingoensis* responded positively

500   (increase in colonisation probability in sites surrounded by pine plantations).

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502

503 Table 1. Model ranking according to ΔAIC (delta Akaike Information Criterion); only models
504 <4 are shown. Ψ = probability of a site being occupied during the first survey, ϒ = probability
505 of colonization; ε = probability of extinction, p = detection probability; Vegtype = vegetation
506 type (categories listed in methods); Y = year (categorical covariate); Yn = year (numeric
507 covariate); H = tree cover within 250 m buffer; nPars = number of estimated parameters; R^2
508 = Nagelkerke's coefficient of determination; (.) = constant model (no covariate).

509

Species	model	nPars	ΔAIC	R^2
Carlia tetradactyla	Ψ(Vegtype)ϒ(T)ε(.)p(Y)	12	0.00	0.46
	Ψ(Vegtype)ϒ(T)ε(Yn)p(Y)	13	1.16	0.47
	Ψ(Vegtype)ϒ(T)ε(H)p(Y)	13	1.96	0.46
	Ψ(Vegtype)ϒ(T)ε(T)p(Y)	13	2.00	0.46
	Ψ(Vegtype)ϒ(T)ε(T*Yn)p(Y)	15	2.65	0.48
	Ψ(Vegtype)ϒ(H)ε(.)p(Y)	12	3.59	0.44
	Ψ(Vegtype)ϒ(T*Yn)ε(.)p(Y)	14	3.64	0.46
	Ψ(Vegtype)ϒ(.)ε(.)p(Y)	11	3.79	0.43
Hemiergis talbingoensis	Ψ(Vegtype)ϒ(T)ε(.)p(Yn)	8	0.00	0.19
	Ψ(Vegtype)ϒ(T)ε(T)p(Yn)	9	1.18	0.20
	Ψ(Vegtype)ϒ(T)ε(Yn)p(Yn)	9	1.80	0.19
	Ψ(Vegtype)ϒ(T)ε(H)p(Yn)	9	1.89	0.19
	Ψ(Vegtype)ϒ(T*Yn)ε(.)p(Yn)	10	3.15	0.20
	Ψ(Vegtype)ϒ(.)ε(T)p(Yn)	8	3.50	0.16

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513   Table S1. Life-history and ecological traits of the Southern Rainbow Skink *Carlia tetradactyla*

514   and the Eastern Three-toed Earless Skink *Hemiergis talbingoensis*. Data was collated from

515   the literature (Greer, 1989; Brown, 1991; Wilson & Swan, 2010).

	Southern Rainbow Skink (*Carlia tetradactyla*)	Three-toed Earless Skink (*Hemiergis talbingoensis*)
Snout-Vent Length (mm)	63	60
Zoogeographical distribution	Bassian	Bassian
Life-form	Terrestrial	Fossorial
Common shelter site	Logs, rocks and leaf Litter	Logs, rocks and debris
Activity patterns	Diurnal	Diurnal but possibly crepuscular and nocturnal (D Michael pers. obs)
Mode of thermoregulation	Heliotherm	Thigmotherm
Mode of reproduction	Oviparous	Viviparous
Clutch/offspring size	2	3
Mating season	Spring	Summer
Oviposition/Neonatal emergence	December/January	February/March
Reproduction frequency/year	1	1
Foraging mode	Active	Active
Dietary preference	Arthropods	Ant/Termites
Operative body temperature	25-32[1]	20
Critical thermal min	*	6.8
Critical thermal max	42	39.3

516   [1]Not available, based on co-generic species (Greer, 1989; Vickers, Manicom & Schwarzkopf,

517   2011)

518   *Not available,

519

520

521 Table S2. Number of sites of the Nanangroe study highlighting the factorial study design

522 (Table modified from Lindenmayer *et al.* 2008).

523

Site	Context	Cohort	No. edges	No. of replicates
Woodland	*Pinus radiata* plantation	1998	1-2	3
Woodland	*Pinus radiata* plantation	1998	3-4	16
Woodland	*Pinus radiata* plantation	2000	1-2	6
Woodland	*Pinus radiata* plantation	2000	3-4	25
Woodland	Grazing land			55

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527 Table S3. Main characteristics of 115 sampled sites. Context: treatment = *Eucalypt* patches

528 surrounded by pine plantation; control = *Eucalypt* patches surrounded by grazing land; pine

529 control = sites within pine plantations. Tree cover (standardized) measured in a 250 m

530 circular buffer surrounding the site. Vegetation type: 1 = red box and red stringybark

531 (codominant) with apple box (*E. bridgesiana*), long-leaf box (*E. goniocalyx*), and broad-

532 leaved peppermint (*E. dives*); (2) mountain swamp gum (*E. camphora*) and other kinds of

533 vegetation (e.g., river oak *Allocausarina cunninghamiana*); (3) yellow box, white box, red

534 stringybark (codominant), and Blakely's red gum.

535

Site code	Experimental design	Tree cover	Vegetation type
AWA-1	Control	-0.162	3
AWA-10	Control	1.412	3
AWA-11	Control	0.709	3
AWA-2	Control	-0.157	3
AWA-3	Control	-0.808	3
AWA-4	Control	0.817	3
AWA-5	Control	0.086	3
AWA-6	Control	-0.014	3
AWA-7	Control	-0.110	3
AWA-8	Control	-0.436	3
AWA-9	Control	0.723	3
GRE-1	Control	-0.007	3
GRE-2	Control	0.133	3
GRE-3	Control	-0.407	3
GRE-4	Control	0.705	3
JWA-1	Control	0.416	3
JWA-10	Control	2.372	3
JWA-11	Control	2.835	3
JWA-2	Control	0.120	3
JWA-3	Control	-0.454	3
JWA-4	Control	0.747	3
JWA-5	Control	1.760	3
JWA-6	Control	-0.256	3
JWA-7	Control	-0.514	3
JWA-8	Control	0.498	3
JWA-9	Control	1.113	3
KEA-1	Control	0.547	3
KEA-2	Control	-0.134	2
KEA-3	Control	0.278	3
KEA-4	Control	1.281	3
KEA-5	Control	0.031	3
KEA-6	Control	-0.367	3

LUF-1	Control	-0.571	3
LUF-10	Control	-0.681	3
LUF-11	Control	-0.846	3
LUF-12	Control	-0.467	3
LUF-13	Control	1.029	3
LUF-14	Control	0.603	3
LUF-2	Control	1.209	3
LUF-3	Control	-0.194	3
LUF-5	Control	-0.646	3
LUF-6	Control	-1.251	2
LUF-7	Control	-0.896	3
LUF-8	Control	-0.593	3
LUF-9	Control	-0.958	3
SKI-1	Control	-0.309	3
SKI-10	Control	1.642	3
SKI-2	Control	-0.451	3
SKI-3	Control	1.498	3
SKI-4	Control	1.262	3
SKI-5	Control	-0.282	3
SKI-6	Control	-0.123	3
SKI-7	Control	0.809	3
SKI-8	Control	1.755	1
SKI-9	Control	2.022	3
PIN-1	Pine control	0.610	6
PIN-10	Pine control	-1.510	6
PIN-2	Pine control	-1.264	6
PIN-3	Pine control	-1.537	6
PIN-4	Pine control	-1.568	6
PIN-5	Pine control	-1.541	6
PIN-6	Pine control	-1.577	6
PIN-7	Pine control	-1.577	6
PIN-8	Pine control	-1.227	6
PIN-9	Pine control	-1.577	6
BUN-1	Treatment	-1.309	3
BUN-2	Treatment	-1.266	3
COT-1	Treatment	0.996	1
COT-10	Treatment	0.725	3
COT-2	Treatment	0.414	1
COT-4	Treatment	-0.833	1
COT-5	Treatment	-0.042	1
COT-6	Treatment	0.074	3
COT-7	Treatment	0.123	3
COT-8	Treatment	0.710	3
COT-9	Treatment	2.004	3
EAB-1	Treatment	1.247	2
EAB-2	Treatment	-0.113	2

|			
EAB-3	Treatment	-1.258	2
EAB-4	Treatment	0.771	2
EAB-5	Treatment	0.009	2
EAB-6	Treatment	1.161	1
EAB-7	Treatment	-0.408	2
EAB-8	Treatment	-0.422	2
NAN-1	Treatment	-1.206	3
NAN-10	Treatment	-0.515	3
NAN-11	Treatment	-0.526	3
NAN-12	Treatment	0.406	3
NAN-14	Treatment	0.520	3
NAN-15	Treatment	-1.089	2
NAN-16	Treatment	-0.819	3
NAN-18	Treatment	1.517	3
NAN-19	Treatment	-0.922	3
NAN-20	Treatment	0.500	3
NAN-23	Treatment	-0.457	1
NAN-24	Treatment	-0.479	1
NAN-25	Treatment	1.255	3
NAN-26	Treatment	-0.135	3
NAN-27	Treatment	0.945	3
NAN-28	Treatment	0.123	3
NAN-29	Treatment	-0.094	2
NAN-30	Treatment	0.363	2
NAN-31	Treatment	-1.041	1
NAN-34	Treatment	0.325	3
NAN-35	Treatment	-1.149	3
NAN-36	Treatment	0.333	3
NAN-37	Treatment	0.362	3
NAN-38	Treatment	0.809	2
NAN-39	Treatment	0.155	3
NAN-4	Treatment	-1.106	3
NAN-40	Treatment	1.554	3
NAN-5	Treatment	-0.618	3
NAN-6	Treatment	-0.682	3
NAN-7	Treatment	1.397	3
NAN-8	Treatment	-0.390	3

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539

540 Fig. S1. Artificial refuges (ARs) used to monitor the presence/absence of target species. ARs
541 composed of: (1) one double-layered stack of corrugated galvanised steel, (2) four 1-m
542 mountain gum *E. dalrympleana*) fence posts covered in mesh, (3) four concrete roof tiles (32
543 cm*42 cm).

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545

546   Fig. S2. Open eucalypt woodland patch (above) and pine plantation (below).

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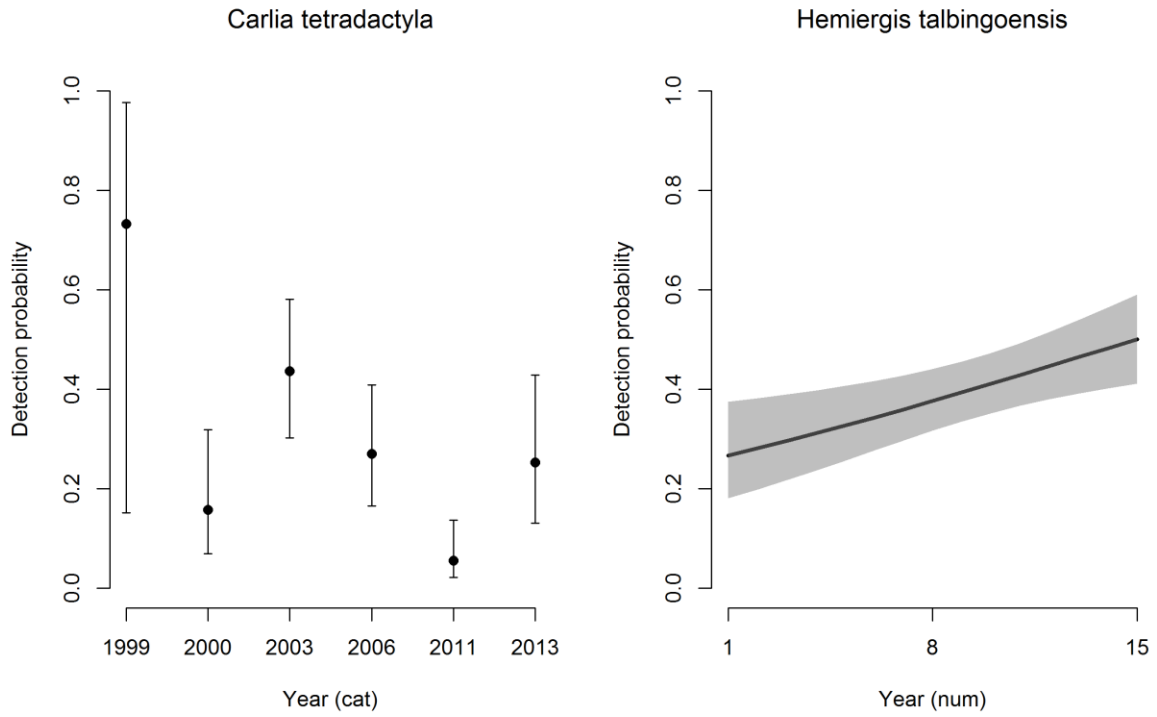
548

549　Fig. S3. Estimates of detection probability for the Southern Rainbow Skink *Carlia*

550　*tetradactyla* and the Eastern Three-toed Earless Skink *Hemiergis talbingoensis*. In *Carlia*

551　*tetradactyla* the probability of detection is year-specific (categorical covariate year),

552　whereas in *H. talbingoensis* the probability of detection increases each year (numerical

553　covariate year). Bars (left) and grey areas (right) are 95% confidence intervals.

554