

1 Australian Forestry

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3 **Diversity and abundance of Lepidoptera and Coleoptera in Greenfleet**
4 **reforestation plantings to offset carbon emissions: Proximity to remnants**
5 **will influence re-wilding of plantings**

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16 **ABSTRACT**

17 Mixed-species (floristically diverse) plantings of trees and shrubs in former agricultural
18 landscapes to offset (sequester) carbon emissions are a recent component of Australian
19 landscapes. Although their potential to mitigate biodiversity loss is recognised, this ecological
20 function has not been investigated, in particular with respect to insect diversity. Over two
21 summers, we used light trapping to sample Lepidoptera (moths) and Coleoptera (beetles) in
22 Greenfleet plantings in two distinct locations in Victoria (plantings of four ages per location)
23 as well as in nearby remnant forest and in pasture. At both locations, we found that plantings
24 had a greater abundance of Lepidoptera than remnants but that the abundance in plantings was
25 comparable to the abundance in pasture. The species richness of Lepidoptera in plantings did
26 not differ significantly from that in remnants but was significantly greater than that in pasture.
27 The abundance and species richness of Coleoptera in plantings was lower than in remnant
28 forests but higher than in pasture. The community composition of Lepidoptera and Coleoptera
29 in plantings was intermediate between that of remnant forest and pasture, i.e. possibly
30 transitional between the two vegetation types. Dissimilarity between all vegetation types was
31 nevertheless high reflecting that the abundance of individual taxa reflects the influence of
32 temporally and spatially dependent factors, e.g. host plant size and suitability. Greater
33 abundances of grass-feeding Lepidoptera in plantings explained much of the dissimilarity
34 between plantings and remnant forests. Proximity to remnant forest was an important
35 determinant of the community composition of Lepidoptera but not of Coleoptera. Moth
36 forewing length (relevant to vagility) appeared less important to proximity relationships than
37 larval host plant specificity, i.e. whether grass- or dicot-feeding. The location of sequestration
38 plantings relative to remnant forest as well as their composition (including the persistence of
39 pasture grasses) will determine the attractiveness of the resources provided to insect herbivores
40 and hence the rate at which they are re-wilded. Greenfleet plantings near remnant native forest

41 benefit insect diversity but adoption of novel silvicultural practices could hasten the rate at
42 which they become functional mimics of native forests and support more comparable
43 communities of insect. Potential trade-offs between increased establishment costs and more
44 complex carbon accounting might need to be investigated to cost biodiversity credits associated
45 with dual accreditation schemes.

46

47 **KEYWORDS**

48 Biodiversity; conservation biology; ecosystem services; restoration ecology

49 **Introduction**

50 Reforestation of long deforested land, such as pasture, is a potential avenue for restoring
51 biodiversity. Nevertheless, most tree planting is for commercial production purposes rather
52 than purely to restore biodiversity. Hence, trees are most often planted as monocultures and
53 any biodiversity benefits (to vertebrates or invertebrates) are coincidental outcomes (Hobbs et
54 al. 2003; Cunningham et al. 2005). The establishment of mixed-species plantings of trees and
55 shrubs represents a significant break from conventional silviculture and reflects changing
56 economic and social values for the use of previously cleared land (Kanowski & Catterall 2010).
57 Recognition of the potential synergies between tree planting for carbon sequestration and
58 biodiversity conservation has resulted in some dual accreditation schemes that enable sponsors
59 to fund reforestation projects (Bekessy & Wintle 2008; Deal et al. 2012). Before biodiversity
60 credits can be offered to sponsors, the capacity of mixed-species sequestration plantings to
61 increase insect diversity needs to be quantified. Such evidence could reassure sponsors of the
62 validity of claims that the insect composition of revegetation plantings is similar to native
63 forests which is the outcome they want to support.

64 Plant growth form (encompasses ‘plant architecture’) and plant apparency are key
65 ecological concepts fundamental to reforestation for the restoration of biodiversity. Trees with
66 their more complex architecture support richer insect faunas than less architecturally complex
67 herbs (Lawton 1983). Such relationships are also apparent within groups of related plants. For
68 example, in a common garden experiment with 21 species of Brassicaceae ranging in height
69 from 10 to 130 cm, Schlinkert et al. (2015) found that larger species of plant supported more
70 species of herbivore, natural enemies and also, but somewhat less so, pollinators. Of the two
71 hypotheses proposed to explain this phenomenon (the size *per se* hypothesis and the resource
72 diversity hypothesis), the resource diversity hypothesis possibly best explains why insect
73 richness increases with plant ontogeny. That is, as an individual plant increases in size as it

74 grows, it provides a greater array of resources (modules and tissues for consumption as well as
75 microhabitats for shelter) for more species of insect herbivore. The trophic significance of this
76 is that a greater diversity of herbivores (second trophic level) supports a more complex
77 community of natural enemies, i.e. the third (predators and parasitoids) and fourth
78 (hyperparasitoids) trophic levels. That such responses are likely to occur as plantation
79 *Eucalyptus* grow has been demonstrated by Steinbauer et al. (2006). Plant growth form and
80 size also determines plant apparency to insect herbivores, i.e. ‘the vulnerability of an individual
81 plant to discovery by its enemies [herbivores]’ (Feeny 1976). Ignoring the putative plant
82 defence syndromes proposed by the apparency hypothesis, recent meta-analyses have found
83 that apparency is confounded with plant life history traits including woodiness and stature
84 (Strauss et al. 2015). Hence, plant lineages differ in the extent of arthropod herbivory they
85 experience and, overall, woody plants experience 64% higher herbivory than non-woody plants
86 (Turcotte et al. 2014). The plant apparency hypothesis has recently been reframed to
87 encapsulate the influence of the searching environment and the host searching abilities of
88 herbivores (Strauss et al. 2015). That is, the sensory (visual and olfactory) and dispersal
89 abilities of herbivores will influence the likelihood of them finding plants when they are in
90 conspicuous and inconspicuous habitats. Prior to Strauss et al. (2015), the influence of
91 apparency and ‘associational resistance’ (*sensu* Tahvanainen & Root 1972) on the utilisation
92 of plantation eucalypts by the autumn gum moth (*Mnesampela privata*) was tested by
93 Steinbauer (2005). Interestingly, Steinbauer (2005) found that isolated host eucalypt (trees
94 surrounded by non-eucalypts) received more moth eggs than trees of the same species growing
95 surrounded by others of the same species, i.e. conspicuous trees were more likely to be found
96 by this host specialist insect than inconspicuous trees. This response was suggested to be
97 exacerbated by the fact that all trees were growing in an ex-pasture situation surrounded by no
98 other vegetation.

99 Insect host plant specificity (or diet breadth) is another key ecological phenomenon that
100 will influence the species richness and eventual community composition of herbivores in
101 sequestration plantings. Most insect herbivores feed upon a limited number of taxonomically
102 (often phytochemically) related species of plant (Novotny et al. 2002; Ødegaard et al. 2005).
103 Fewer species are monophagous [i.e. ‘a species with at least 90% of individuals feeding on a
104 single host species’ (Novotny & Basset 2005)] or polyphagous (feeding on more than one plant
105 family). Consequently, different species of plant in the same habitat may share some insect
106 herbivores in common with one another but will also host a unique suite of species (Forbes et
107 al. 2017). As a result, the diversity of invertebrates in a given habitat will mirror plant species
108 richness (Castagneyrol & Jactel 2012). Relatedly, insect richness has been found to track
109 successional (temporal) changes in plant richness (Lewinsohn et al. 2005). In a light trapping
110 study of Lepidoptera in deciduous forest in Ohio, host specialist (or diet-restricted) moths
111 contributed less to stand-level diversity than host generalist moths (Summerville et al. 2006).
112 The re-wilding of sequestration plantings by insect herbivores is also likely to be influenced
113 by body size (related to vagility and hence dispersal capacity) and landscape context. A
114 comparative study of geometrid moths has shown that larger species have broader host plant
115 ranges than smaller species (Davis et al. 2013). Research using flight interception traps reported
116 that older restoration plantings and those adjacent to rainforest had a higher rainforest-like
117 beetle composition than those > 0.9 km from remnants (Grimbacher & Catterall 2007). A
118 similar finding but relating to Coleoptera, Lepidoptera and Hymenoptera was not reported by
119 Cunningham et al. (2005) in relation to commercial (monoculture) plantations of *Eucalyptus*
120 *globulus*.

121 For this study we light trapped for Lepidoptera and Coleoptera. We used light trapping
122 because we were particularly interested in studying insects likely to be significant defoliators
123 of the eucalypts planted by Greenfleet (<https://greenfleet.com.au/Home>). The immature life

124 cycle stages of such insects either cannot be collected from foliage (e.g. the larvae of scarab
125 beetles) or require extensive sweep-netting, beating and/or insecticide fogging of foliage (e.g.
126 larval Lepidoptera) to sample adequately. Since adults are generally strong fliers, they are more
127 readily sampled by various types of attractive trap, e.g. light traps. Our application of light traps
128 to this study is somewhat unique in that numerous other Australian studies use pitfall (majority
129 of studies) or flight interception traps or fogging – none of which act by attraction of insects.
130 Using Greenfleet plantings, our aims were to: (1) compare insect abundance and species
131 richness in plantings, remnant forests and pasture (2) compare insect composition in plantings,
132 remnant forests and pasture and (3) examine the importance of proximity to remnant forest
133 (landscape context) to selected abundant species.

134 **Materials and methods**

135 **Study areas**

136 This study was conducted in carbon offset plantings established by Greenfleet in areas
137 comprising a mosaic of remnant forest and pasture. Greenfleet works with private landholders,
138 local and State Governments when reforesting cleared land, i.e. cleared prior to 1990.
139 Greenfleet plants a mix of native species that would have been present in an area prior to land
140 clearing. In Victoria, plants for reforestation are grown from locally sourced seed which is
141 harvested by either Parks Victoria or local nurseries, e.g. Smolders Revegetation
142 (<http://www.smoldersrevegetation.com.au/>) in the case of the areas we studied. Our two study
143 areas were in Mount Worth State Park and Devilbend Reserve (Fig. 1). The Bureau of
144 Meteorology (BoM) climate data most relevant to Devilbend Reserve comes from Mornington
145 (station number 086361; mean annual maximum & minimum temperatures 18.9°C & 10.1°C,
146 mean annual rainfall 739.7 mm and height 60 m) and to Mount Worth State Park comes from
147 Erica (station number 085026; mean annual maximum & minimum temperatures 16.7°C &
148 7.9°C, mean annual rainfall 1,103.5 mm and height 440 m). The remnant forest at Mount Worth
149 State Park is wet sclerophyll dominated by *Eucalyptus regnans* while the remnant vegetation
150 at Devilbend Reserve is coastal forest dominated by *E. viminalis*, *E. radiata* and *E. ovata*. In
151 each area, we sampled from four Greenfleet plantings (representing four planting dates), four
152 remnant forest sites and two pasture sites (Table 1). We attempted ‘space-for-time’ substitution
153 and pairing of plantings with remnants but each planting was reforested with different
154 combinations of species and most had been established within a relatively short period of time
155 of each other (providing limited spread of ages). Greenfleet plantings had been established in
156 the pasture vegetation in which we trapped for insects. The age of the plantings we were able
157 to sample from was constrained by the history of reforestation in each area. The proximity of
158 each light trap to the closest remnant was estimated using ArcGIS version 10.2.2.

159

160 **Insect sampling**

161 Light trapping was conducted over two summers, i.e. December, January and February of
162 2014-2015 and 2015-2016. We sampled insects from each site for one night per month. Sites
163 in Mount Worth State Park were sampled from December 2014-February 2016 (a total of 6
164 sampling nights) while sites in Devilbend Reserve were sampled from February 2015-February
165 2016 (a total of 4 sampling nights). One light trap equipped with a vertical 8 W ultraviolet tube
166 (Australian Entomological Supplies Pty. Ltd., Bangalow, New South Wales) was placed at
167 each site in the same location (marked by a half-length star picket) throughout the sampling to
168 collect moths (Lepidoptera) and nocturnal beetles (Coleoptera). We timed our light trapping to
169 occur within three nights of the new moon to minimise the negative effect of light competition
170 on trap catch (Steinbauer 2003; Steinbauer et al. 2012). Moon phase data were obtained from
171 the U.S. Naval Observatory, Astronomical Applications Department website
172 (<http://aa.usno.navy.mil/data/docs/MoonPhase.php>). Pieces of egg carton (to provide refugia)
173 and a vial of ethyl acetate (to speed knockdown of specimens) were placed inside each trap.
174 Light traps came on automatically at dusk and turned off automatically at dawn. Specimens
175 were removed from traps the following morning, stored in 750 mL plastic Décor containers
176 lined with tissues and kept in a car freezer prior to returning to La Trobe University. Specimens
177 were stored at -18°C until they could be sorted, identified and enumerated.

178 Despite our attempts to preserve the condition of moths, specimens caught in bucket
179 light traps are often damaged, especially when many insects are caught in the same night.
180 Consequently, small moths ('microlepidoptera') could not be reliably identified as
181 morphospecies let alone to species level so only 'macrolepidoptera' (i.e. body length ≥ 10 mm)
182 were enumerated. Only beetles with body length ≥ 5 mm were identified and enumerated.

183 Lepidoptera were identified to species level using the *Moths of Victoria* (Parts 1 to 8 and
184 including the online resources) followed by later expert examination (see Acknowledgements).
185 Coleoptera were identified by comparison to specimens in the private collection of MJS
186 (identified previously by Tom Weir; Steinbauer & Weir 2007) and using online resources.
187 Specimens unable to be identified to species level were identified using a morphospecies
188 approach. The species of Lepidoptera and Coleoptera trapped are listed in Appendices 1 and 2,
189 respectively. Voucher specimens are lodged in the insect collection of the Department of
190 Ecology, Environment and Evolution. Forewing lengths of selected species of Lepidoptera
191 were measured using an electronic caliper. The flight wings of beetles of interest were not
192 measured because they would have to be removed from beneath the elytra.

193

194 **Statistical analyses**

195 We ran generalised linear mixed models (GLMMs) using package lme4 (Bates et al. 2015) in
196 R to test differences in insect abundance and species richness between plantings, remnant
197 forests and pasture. For each study area, sampling periods were regarded as repeated measures.
198 We did not examine differences between each year but instead pooled the data. Site was
199 included as random effect and vegetation type was a predictor variable. Data were log-gamma
200 transformed so that their distribution was normal.

201 An NMDS ordination was produced in R to compare the community composition of
202 insects in plantings, remnant forests and pasture. We excluded “unknown” species and those
203 species present on fewer than four occasions across all sampling events. Figures were created
204 using the ‘ggplot2’ package for R.

205 The insect composition of plantings, remnants and pasture were examined using
206 similarity percentage (SIMPER) analyses on log+1 transformed data in PRIMER-E Version 7.

207 SIMPER decomposes Bray-Curtis similarities between all pairs of samples to identify species
208 that contribute most to the differences observed. We were interested in examining whether
209 there were differences in the abundance of different feeding behaviours among insects in
210 plantings, remnant forests and pasture, however, given the large number of species collected
211 (and lack of knowledge about the ecology of many of them) we examined only the 40% most
212 frequently occurring species in our light traps. Cut-off contributions were set at 70%.

213 To assess the influence of proximity to remnant forest on the composition of
214 Lepidoptera and Coleoptera we used multivariate generalised linear models (GLMs) using the
215 manyglm function in the R package mvabund (Wang et al. 2012). We used a negative binomial
216 distribution and untransformed abundances to analyse catches of different species relative to
217 the proximity to remnant forest.

218 **Results**

219 **Insect abundance and species richness**

220 Over the two summer when light trapping was conducted, we identified 253 species of
221 Lepidoptera (represented by 2,782 individuals) in remnants, 205 species (represented by 4,129
222 individuals) in plantings and 78 species (represented by 1,245 individuals) in pasture (Table
223 2). Despite the large overall differences in abundance, we did not find any difference in the
224 mean abundance of Lepidoptera at individual sites between pasture and plantings (GLMM,
225 estimate = 0.418, $t = 1.51$, SE = 0.276, $P = 0.130$) or between pasture and remnant forests
226 (GLMM, estimate = 0.013, $t = 0.045$, SE = 0.288, $P = 0.964$). However, the mean abundance
227 of Lepidoptera at sites was significantly greater in plantings than in remnant forests (GLMM,
228 estimate = -0.456, $t = -69.5$, SE = 0.007, $P < 0.001$). Noctuidae (cutworm or armyworm family)
229 was the most abundant family across all site types. The next most abundant family in plantings
230 and remnants was Hepialidae (swift moth or ghost moth family) and Geometridae (geometer
231 family, including emerald moths of the subfamily Geometrinae), respectively.

232 The species richness of Lepidoptera in pasture was significantly lower than in plantings
233 (GLMM, estimate = 0.574, $t = 2.24$, SE = 0.255, $P < 0.025$) and in remnants (GLMM, estimate
234 = 0.790, $t = 4.12$, SE = 0.192, $P < 0.001$) whereas the species richness of Lepidoptera in
235 plantings and remnants did not differ significantly (GLMM, estimate = 0.232, $t = 1.048$, SE =
236 0.222, $P = 0.295$). Geometridae was the most species rich family in remnants while Noctuidae
237 was the most species rich family in both plantings and pasture.

238 We trapped a number of undescribed moth species. At Mount Worth we trapped
239 specimens of two undescribed species of *Austroterpna* Goldfinch (sp. 1 and sp. 2; Geometridae;
240 plantings and remnants), an undescribed species of *Chlorocoma* Turner (sp. 1; Geometridae;
241 plantings and remnants) and an undescribed species of *Furcatrox* McQuillan (sp. 1;

242 Geometridae; remnants only). We also trapped *Chrysolarentia euphileta* (Turner)
243 (Geometridae; plantings, remnants and pasture) at Mount Worth. As at June 2015, *C. euphileta*
244 had only previously been recorded from Otway National Park (Peter Marriott, pers. comm., 3
245 June 2015). At Devilbend, we trapped specimens of an undescribed species of *Monoctenia* (sp.
246 1; Geometridae; remnants only). The undescribed species remain to be formally described.

247 Over the same period, we identified 193 species of Coleoptera (represented by 7,218
248 individuals) in remnants, 137 species (represented by 8,935 individuals) in plantings and 36
249 species (represented by 454 individuals) in pasture (Table 2). The abundance of Coleoptera
250 was significantly lower in pasture compared to plantings (GLMM, estimate = 2.11, $t = 4.68$,
251 SE = 0.451, $P < 0.001$) and remnant forests (GLMM, estimate = 1.87, $t = 5.12$, SE = 0.365, P
252 < 0.001). Abundance did not differ significantly between plantings and remnant forests
253 (GLMM, estimate = -0.238, $t = -0.760$, SE = 0.313, $P = 0.447$). Catches in planting, remnant
254 and pasture sites were dominated by beetles in the family Scarabaeidae (scarab family).

255 The species richness of pasture was significantly lower than plantings and remnants
256 (GLMM, estimate = 0.747, $t = -2.61$, SE = 0.286, $P = < 0.01$; estimate = 1.19, $t = 4.63$, SE =
257 0.258, $P < 0.001$). The species richness of Coleoptera in plantings was significantly lower than
258 in remnant forests (GLMM, estimate = 0.446, $t = 2.07$, SE = 0.215, $P < 0.05$). Scarabaeidae
259 was the most species rich family across all sites. The next most species rich families in
260 plantings and remnants were Elateridae (click beetle family) and Cerambycidae (longhorn
261 beetle family).

262

263 **Insect composition**

264 In NMDS similarity space, the communities of both Lepidoptera (Fig. 2a) and Coleoptera (Fig.
265 2b) in plantings were intermediate in composition between remnant forest and pasture. For

266 both orders at each location, the composition of the communities at individual sites were
267 different from one another (Fig. 2). For Lepidoptera, the communities in plantings at Mount
268 Worth (squares) were more similar to those in remnants than they were to the communities of
269 moths in plantings and more so than the communities of moths at Devilbend (circles; Fig. 2).
270 The composition of both Lepidoptera and Coleoptera in ‘older’ plantings was more similar to
271 the composition in remnant forests while the composition in ‘younger’ plantings was more
272 similar to pasture.

273 The community composition of Lepidoptera in plantings and remnants differed slightly
274 less (SIMPER, dissimilarity = 74.1%) than did the composition of moths in plantings and
275 pasture (SIMPER, dissimilarity = 75.6%). Remnant forests and pasture had the greatest
276 dissimilarity (SIMPER, dissimilarity = 79.8%). Species contributing most to the differences
277 between vegetation types were grass-feeding including some agricultural pest species, e.g.
278 *Hednota pleniferellus* (Crambidae), *Proteuxoa sanguinipuncta* (Noctuidae) and *Persectania*
279 *ewingii* (Noctuidae) (Table 3). Pasture and plantings typically had greater abundances of grass-
280 feeding species compared to remnants. Much of the dissimilarity between pasture and remnant
281 forests was due to greater abundances of eucalypt- and acacia-feeding species in remnants
282 (Table 3).

283 Of the three vegetation types, the community composition of Coleoptera in plantings
284 and remnants were most similar to each other (SIMPER, dissimilarity = 71.6%) (Table 3). The
285 community composition of beetles in pasture more closely resembled remnant forests
286 (SIMPER, dissimilarity = 79.5%) than of plantings (SIMPER, dissimilarity = 82.8%). *Telura*
287 *vitticollis* (Scarabaeidae) and *Sericesthis nigrolineata* (sp. 33) (Scarabaeidae) combined were
288 responsible for more than 30% of the differences between each vegetation type (Table 3).
289 Specimens subtly different from *Sericesthis nigrolineata* (sp. 33) but also identified as the same
290 species were trapped (Table 3); these specimens were identified as *S. nigrolineata* (sp. 41) and

291 *S. nigrolineata* (sp. 42). As currently recognised, *Sericesthis nigrolineata* is a common,
292 widespread and phenotypically variable species (Tom Weir, pers. comm., 26 June 2018).

293

294 **Importance of proximity to remnant forest on community composition**

295 The multivariate glm analysis using the abundances of the most frequently occurring species
296 as the response variable indicated that Lepidoptera were significantly influenced by proximity
297 to remnant forest (GLM, deviance = 105.7, $P = 0.009$) whereas Coleoptera were not (GLM,
298 deviance = 33.3, $P = 0.182$). Eleven species of Lepidoptera (out of 22 used in analyses) were
299 significantly influenced by proximity to remnant forest; seven were negatively influenced by
300 proximity to remnant forest and four were positively influenced (Table 4). Three species of
301 Coleoptera were significantly (negatively) influenced by proximity to remnant forest. The
302 forewing lengths of Lepidoptera the abundances of which were influenced (positively or
303 negatively) by proximity to remnant forest exhibited no obvious patterns with respect to size,
304 e.g. smaller wings in species negatively associated with distance to remnant and *vice versa*
305 (Table 5). Increased distance from remnant forest had a negative influence on catches of
306 foliage-feeding Lepidoptera whereas it had a positive influence on catches of grass-feeding
307 moth species (Table 4).

308 **Discussion**

309 We present the findings of the first light trap study to be conducted in mixed-species plantings
310 of trees and shrubs established to offset (sequester) carbon emissions. Ours is only the second
311 study in Australia to use light traps to compare insect diversity in plantings, remnant forest and
312 pasture; the first was the study by Cunningham et al. (2005). We recorded greater species
313 richness of Lepidoptera and Coleoptera in plantings compared to pasture which demonstrates
314 the ability of Greenfleet's mixed-species forests to increase insect diversity. The use of almost
315 any sampling technique (e.g. light traps, coloured or baited traps, pitfall traps, Malaise traps)
316 will bias the representation of insect taxa sampled (Steinbauer et al. 2012). Experimental
317 studies of the attractiveness of light traps to Australian insects are none existent. Steinbauer et
318 al. (2001) reported catching a gravid autumn gum moth [*Mnesampela privata* (Geometridae)]
319 in a 20 W ultraviolet light trap that was 224 m from the nearest host eucalypt and that
320 experimental plantings of eucalypt hosts separated by 240 m of open pasture were colonised
321 by this individuals of this species within one month of each other. Using mark-release-
322 recapture, Östrand & Anderbrant (2003) estimated that 50% of male European pine sawflies
323 [*Neodiprion sertifer* (Hymenoptera: Diprionidae)] originated up to 450 m from sex pheromone
324 traps (also an attractant type of trap). Consequently, the immature stages of the insects we
325 trapped presumably developed on host plant(s) either in the same habitat as our light traps or
326 in abutting habitats.

327 **Does insect abundance and species richness differ among plantings, remnant forests and**
328 **pasture?**

329 Plantings had a greater abundance of Lepidoptera compared to remnant forests, largely due to
330 high abundances of Noctuidae (notably *Agrotis* and *Persectania* species), Hepialidae
331 (*Abantiades labrinthicus* and *Elhamma australasiae*) and Crambidae (*Hednota* species). These

332 species are primarily grass-feeding and a couple are considered agricultural pests of crops and
333 pasture, e.g. *Agrotis* and *Persectania*. Younger Greenfleet plantings were characterised by a
334 grassy understorey which likely explains the high abundance of grass-feeding species of
335 Lepidoptera and Coleoptera (especially Scarabaeidae). Cunningham et al. (2005) suggested
336 that commercial *E. globulus* plantations may facilitate the movement of forestry pests (e.g. the
337 eucalypt herbivore *Gonipterus scutellatus sensu lato*) into nearby remnant forests. In addition
338 to higher abundances of pest moth species in plantings, we trapped 207 *Proteuxoa*
339 *sanguinipuncta* at one remnant site at Mount Worth (MW10R) in a single night in January
340 2016. Such events are likely explained by the vagility of many of these moths and perhaps the
341 invasion of pasture grass species someway into remnants. In commercial plantations, grasses
342 are controlled for 1-2 years after establishment using herbicide and later the dense planting and
343 rapid growth of trees shades out the remaining grass (Adams et al. 2003). In the absence of
344 such silvicultural management, sequestration plantings could continue to provide resources for
345 grass-feeding species which might otherwise not occur in an earlier successional forest. Among
346 grass-feeding beetles, those in the subfamily Melolonthinae (including the genera *Automolius*,
347 *Heteronychus*, *Heteronyx* and *Liparetrus*) can be serious threats to seedling eucalypts
348 (Steinbauer & Weir 2007).

349 Greenfleet plantings hosted communities of Lepidoptera as diverse as remnant forests.
350 These findings differ from studies examining insect communities in single-species
351 (monoculture) plantings which reported lower insect diversity compared to remnant forests
352 (Hobbs et al. 2003; Cunningham et al. 2005; Robson et al. 2009) but are in agreement with the
353 results of studies comparing mixed-species plantings with remnants (Dunn 2004; Moir et al.
354 2005; Grimbacher et al. 2007). This trend is also in agreement with that reported for species of
355 native wasp parasitoids in plantations of differing levels of tree diversity (Steinbauer et al.
356 2006). Central to this result is the proximity to remnants (see next) and the efficacy of using

357 light traps to sample Lepidoptera. That is, as a taxon, Lepidoptera is dominated by families that
358 are entirely or largely nocturnal. Hence, light traps are likely to sample a broader suite of taxa
359 of Lepidoptera than of Coleoptera.

360 The species richness of Coleoptera in plantings was significantly lower than in remnant
361 forest which might suggest that Coleoptera require longer to re-wild plantings than do
362 Lepidoptera. However, and ignoring the influence of the probable bias in beetle taxa sampled
363 as a consequence of using light traps (in favour of Scarabaeidae, especially Melolonthinae),
364 explanation of our finding probably has multiple components. Few general trends relating to
365 the importance of dispersal capability alone on beetle responses to habitat fragmentation have
366 been reported for Australian taxa (Driscoll & Weir 2005 *cf.* Cunningham & Murray 2007);
367 rather combinations of life history traits have been suggested to explain beetle responses
368 (Driscoll & Weir 2005). Surprisingly, Driscoll (2005) reported that beetle communities in
369 Tasmanian rainforest and eucalypt forest overlapped substantially and therefore proposed that
370 they represented continuous habitat for most species. The Greenfleet plantings had a relatively
371 open canopy and dry, grass dominated understorey which contrasted with the damp conditions
372 of the understorey of the remnants, most notably at Mount Worth. Soil moisture has been
373 shown to alter the community composition of subterranean and epigaeic beetles (Butterfield et
374 al. 1995; Baker 2006; Niemelä et al. 2012). Whether the taxa of beetles readily sampled using
375 pitfall traps are represented comparably in light trap catches does not appear to have been
376 investigated but is considered unlikely.

377 **Does insect composition differ among plantings, remnant forests and pasture?**

378 Not surprisingly, given the relatively young age of Greenfleet plantings, appreciable
379 differences in the community composition of Lepidoptera and Coleoptera between plantings
380 and remnant forests were apparent. Within Lepidoptera, most of the dissimilarity between

381 plantings and remnant forests was due to greater abundances of grass-feeding species in
382 plantings. Generally, our findings are in agreement with previous studies that have reported
383 that invertebrate species richness in restored forests can closely resemble that in remnant forests
384 but community composition does not (Moir et al. 2005; Grimbacher et al. 2007) and is
385 consistent with the suggestion that species richness is the most easily restored component of
386 biodiversity whereas attaining a species composition similar to remnant forests is harder to
387 achieve (Dunn 2004; Valtonen et al. 2017).

388 Our ordination analysis indicates that the community composition of moths and beetles
389 in plantings is transitional between a pasture and a remnant forest community. Our SIMPER
390 analysis also indicates that the compositions of Lepidoptera and Coleoptera in plantings and
391 remnant forests were more similar to one another other compared to that of plantings and
392 pasture. Both finds are positive from a reforestation perspective and indicate that the insect
393 composition of mixed-species plantings is transitioning away from that of pasture and
394 progressing towards remnant forest. Nevertheless, it is difficult to untangle which aspects of
395 site age could be driving insect composition using our data because vegetation and habitat
396 variables could not be assessed thoroughly with the modest resources available to this project.
397 For example, without detailed surveys of the species of plant present in plantings and their
398 relative abundance, we cannot separate the effects of species richness and functional
399 significance on insect composition, e.g. representation of *C*₄ *versus* *C*₃ grasses and/or nitrogen-
400 fixing species of plant (Haddad et al. 2001).

401 As we also expected, there was a clear distinction between the composition of moths
402 and beetles at Devilbend Reserve and the composition of these insects at Mount Worth State
403 Park. Furthermore, the composition of Lepidoptera among sites at Mount Worth State Park
404 were more similar to each other compared to those at Devilbend where the landscape has been
405 more highly fragmented. A light trap study of geometrid moths endemic to Mount Kilimanjaro

406 reported that elevation (1,200-3,150 m) was a better direct predictor of species richness than
407 any index of floristic diversity and structure (Axmacher et al. 2009). The implication of this
408 finding is relevant to our study given the differences in elevation of the two study locations,
409 i.e. 77-87 m at Devilbend and 378-476 m at Mount Worth. The compositional differences
410 between our two locations highlight the importance of specific biodiversity surveys in, for
411 example, dryland or higher rainfall reforestation plantings rather than trying to draw inferences
412 about insect responses based on studies from markedly different habitats. Such considerations
413 would be especially important when trying to infer responses of rare species.

414 **Importance of proximity to remnant forests on insect communities**

415 The proximity of Greenfleet plantings to remnant forest influenced the composition of
416 communities of Lepidoptera and Coleoptera in plantings. We identified seven species of moth
417 (representative for six families) that were less likely to be caught further away from remnants
418 and four other species (representative of three families) that were more likely to be caught
419 further away from remnants. We identified three species of beetle (all Scarabaeidae:
420 Melolonthinae) that were less likely to be caught further away from remnants. It is reasonable
421 to assume that the species listed in Table 4 are modest to strong fliers. Vagility (the capacity to
422 disperse) can be integral to how insects re-colonise hosts in new and isolated locations. Our
423 collection of pinned moth reference specimens can be used to obtain morphometric wing
424 measurements in the future but live specimens are needed to acquire body weight data;
425 measuring wing lengths of beetles would require removing elytra (Jones et al. 2016).
426 Nevertheless, neither wing size (e.g. forewing length – see Table 5) nor wing loading (i.e. body
427 mass divided by wing area) alone can satisfactorily explain dispersal capability because they
428 do not provide sufficient insight into a species' flight physiology and behaviour (e.g. wing
429 muscle thermogenesis and basking) – let alone the stimuli that initiate or sustain host searching.
430 For example, Slade et al. (2013) found that a range of life history traits predicted the mobility

431 of forest macrolepidoptera but that the predictive power of wingspan and wing shape depended
432 upon a species' affinity for unique characteristics of contiguous and expansive forest. Such
433 interactions are suggested to explain why larval host specificity (grass- or dicot-feeding) may
434 better explain adult occurrence in relation to proximity to remnants than forewing length.
435 Moreover, active dispersal of winged adults can be wind assisted while the larvae of some taxa
436 (e.g. some moths) are dispersed passively by wind. For example, variation in the colonisation
437 of Brussel sprouts by *Brevicoryne brassicae* (cabbage aphid) was best explained by the
438 downwind area of crops while variation in colonisation by *Aleyrodes proletella* (cabbage
439 whitefly) was best explained by the upwind area of crops (Ludwig et al. 2018). Interestingly,
440 although *A. proletella* was found to be transported downwind to hosts, *B. brassicae* located
441 hosts during upwind (active) flight undertaken over approximately 1 km. There is clearly
442 ongoing need for 'landscape entomology' (*sensu* Lundquist & Reich 2014) in Australia given
443 the diversity of organisations planting trees and shrubs. Such research has not been conducted
444 since the study by Clarke et al. (1997) in commercial eucalypt plantations.

445 **Conclusions**

446 Mixed-species (floristically diverse) sequestration plantings increase insect diversity primarily
447 because they provide hosts often long since removed from the landscape and which also have
448 more complex plant growth forms and more diverse resources than grasses, i.e. trees and
449 shrubs. Nevertheless, we found high overall dissimilarity in moth and beetle communities in
450 plantings compared to remnants. To explain such dissimilarity requires that functional
451 relationships between resource availability and insect diversity/abundance are quantified.
452 Better knowledge of the life history traits (e.g. Slade et al. 2013; Jones et al. 2016) and identity
453 of the host(s) of specific herbivores would improve our ability to identify insects sustained by
454 a given habitat. Only direct (hand) collection from plants and rearing of immatures, as
455 undertaken by Novotny et al. (2002, 2007), can provide the detailed insect-plant association(s)

456 information required to comprehensively understand how habitat resources sustain insect
457 diversity. Unfortunately, this approach is time consuming and hence more expensive. Direct
458 collection and rearing of immatures, together with vegetation surveys, is recommended for
459 future studies which seek to understand species turnover in response to the aging of plantings.
460 If this is not possible, future insect surveys could utilise sampling methods that are better suited
461 to less vagile taxa and life cycle stages, e.g. sweep netting, beating and/or vacuum sampling. It
462 should be appreciated that sampling techniques such as these will typically have a bias towards
463 smaller insect taxa, e.g. Hemiptera. Ideally also, studies such as ours will provide far greater
464 insight if results are able to be derived from plantings for which factors such as age of planting,
465 species planted and proximity to remnants are better controlled and standardised.

466 We suggest that organisations planting forests to benefit native insects should focus on
467 the development of understorey vegetation and minimisation of the persistence of exotic
468 pasture grasses. Consequently, the adoption of conventional silvicultural practices used for
469 establishing commercial eucalypt plantations needs to be re-considered for the establishment
470 of carbon sequestration plus biodiversity plantings. Specifically, the use of rows (when planting
471 seedlings or direct seeding) and fixed distance spacing (3 m) between trees and shrubs will
472 increase the length of time before canopy closure is achieved thereby increasing the length of
473 time before there is significant shading and retention of moisture at ground level. We suggest
474 that broad-scale herbicide removal of grasses, surface tilling and aerial seeding (using a mix of
475 understorey and/or tree species potentially delivered from drones), possibly followed by some
476 planting of seedlings, will be a more effective way to rapidly and irregularly re-establish tree
477 and shrub hosts and, concomitantly, environmental conditions attractive to a wider suite of
478 insect species than conventional silvicultural practices. Reducing the length of time that pasture
479 grasses persist in reforestation plantings should also reduce the abundance and the potential for
480 mortality of young trees caused by the adults of dicot-feeding scarab beetles as well as reducing

481 the adverse effects of competition. In the absence of knowing the hosts of endemic insects,
482 reforestation projects should also aim to re-establish all plant species formerly endemic to a
483 given location (Moir et al. 2010). We suggest that decreasing the length of time until
484 sequestration plantings become ‘functional mimics’ (*sensu* Lefroy & Hobbs 1998) of native
485 forests will hasten the rate at which the community composition of insects in plantings
486 approaches that of native forest. Since this may be a more expensive way to reforest former
487 agricultural land and could make carbon accounting more challenging, the price of biodiversity
488 credits may need to reflect these additional silvicultural costs. Fortunately, however, native
489 forest remnants are free sources of endemic insect species that will re-wild plantings of their
490 own accord and/or wind assisted in the majority of cases. Only flightless and monophagous
491 specialists (those potentially threatened by coextinction) might need human intervention to
492 colonise plantings, e.g. via translocation of individuals and/or establishment of hosts.

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502

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505

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512

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647 Figure legends

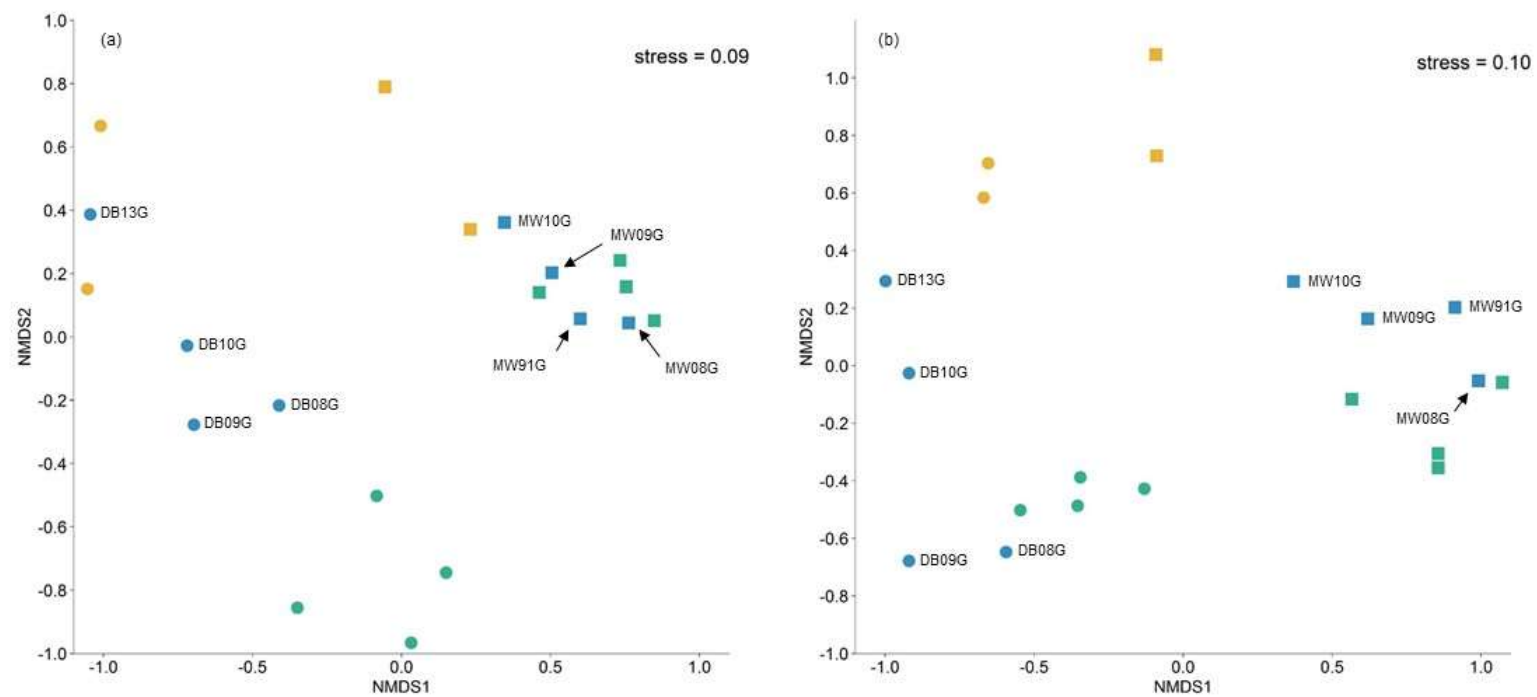
648 **Figure 1.** Maps of study sites at Mount Worth State Park (top) and Devilbend Reserve (bottom)
649 Inset map (top left) of southern Victoria shows relative location of the two study areas. Key to
650 colouration: green areas = native vegetation in 2005; blue areas = water (Devilbend Reservoir);
651 white areas = cleared. Native Vegetation – Modelled 2005 Ecological Vegetation Classes (with
652 Bioregional Conservation Status) layer from [https://www.data.vic.gov.au/data/dataset/native-](https://www.data.vic.gov.au/data/dataset/native-vegetation-modelled-2005-ecological-vegetation-classes-with-bioregional-conservation-status)
653 [vegetation-modelled-2005-ecological-vegetation-classes-with-bioregional-conservation-](https://www.data.vic.gov.au/data/dataset/native-vegetation-modelled-2005-ecological-vegetation-classes-with-bioregional-conservation-status)
654 [status](https://www.data.vic.gov.au/data/dataset/native-vegetation-modelled-2005-ecological-vegetation-classes-with-bioregional-conservation-status). Note: EVC mapping potentially inaccurate at fine scales due to resolution of data and
655 layer having been produced by modelling, e.g. MWP1 and MWP2. Code to study sites given
656 in Table 1.

657

658 **Figure 2.** NMDS ordinations showing the relative similarities in composition of Lepidoptera
659 (a) and of Coleoptera (b). Key to symbols: squares = Mount Worth State Park; circles =
660 Devilbend Reserve; blue symbols = Greenfleet plantings; green symbols = remnant forest;
661 yellow symbols = pasture. Stress < 0.1 indicates that a two-dimensional representation of data
662 is acceptable. Samples closer to each other have higher similarity in species composition. Code
663 to Greenfleet plantings given in Table 1.



666 **Figure 2**



667

668 **Table 1.** Locations of Greenfleet plantings, remnant forest and pasture sites where light trapping was conducted during the summers of 2014-2015
 669 and 2015-2016. One light trap was used at each site in each month of trapping

Greenfleet planting or pasture	Year planted	Code	No. species planted	Latitude & longitude	Elevation (m)	Remnant	Code	Latitude & longitude	Elevation (m)	Month/year of trapping ^a
Mt Worth	1991	MW91G	No record	38°16'30.0"S, 145°58'13.9"E	437	Mt Worth	MW91R	38°17'12.1"S, 145°58'08.0"E	446	1, 2, 3, 4, 5, 6
Mt Worth	2008	MW08G	1	38°17'33.4"S, 145°58'22.8"E	476	Mt Worth	MW08R	38°17'13.8"S, 145°58'06.9"E	447	1, 2, 3, 4, 5, 6
Mt Worth	2009	MW09G	16	38°16'20.2"S, 146°00'43.2"E	425	Mt Worth	MW09R	38°16'32.4"S, 146°00'25.9"E	417	1, 2, 3, 4, 5, 6
Mt Worth	2010	MW10G	15	38°15'55.9"S, 146°00'29.9"E	408	Mt Worth	MW10R	38°16'20.7"S, 146°00'30.2"E	424	MW10G: 1, 2, 3, 4, 5, 6 MW10R ^b : 1, 3, 4, 5, 6
Pasture 1	-	MWP1	-	38°16'34.8"S, 146°00'45.8"E	378	-	-	-	-	1, 2, 3, 4, 5, 6
Pasture 2 ^b	-	MWP2	-	38°16'31.4"S, 146°00'44.9"E	379	-	-	-	-	1, 3, 4, 5, 6
Devilbend ^c	2008	DB08G	14	38°17'47.3"S,	85	Devilbend	DB08R	38°17'43.5"S,	79	3, 4, 5, 6

				145°06'02.5"E				145°06'13.9"E		
Devilbend ^c	2009	DB09G	8	38°17'40.2"S,	80	Devilbend	DB09R	38°17'39.6"S,	87	DB09G ^d : 3, 4, 6
				145°05'48.4"E				145°05'33.7"E		DB09R: 3, 4, 5, 6
Devilbend ^c	2010	DB10G	11	38°18'14.9"S,	82	Devilbend	DB10R	38°17'57.0"S,	81	3, 4, 5, 6
				145°07'32.9"E				145°06'51.0"E		
Devilbend ^c	2013	DB13G	17	38°18'27.5"S,	80	Devilbend	DB13R	38°18'27.5"S,	77	3, 4, 5, 6
				145°06'39.1"E				145°06'26.0"E		
Pasture 1 ^c	-	DBP1	-	38°18'20.0"S,	79	-	-	-	-	3, 4, 5, 6
				145°06'43.8"E						
Pasture 2 ^c	-	DBP2	-	38°18'17.0"S,	79	-	-	-	-	3, 4, 5, 6
				145°06'44.7"E						

670 ^a Key: 1 = December 2014 (first summer), 2 = January 2015 (first summer), 3 = February 2015 (first summer), 4 = December 2015 (second
671 summer), 5 = January 2016 (second summer) and 6 = February 2016 (second summer).

672 ^b No catches from January 2015 due to two malfunctioning light traps.

673 ^c Trapping unable to be conducted in Devilbend Reserve in December 2014 and January 2015 due to delay in issue of collecting permit.

674 ^d Planting not sampled in January 2016 due to light trap malfunction.

675 **Table 2.** Summary of diversity (species richness) and abundance of Lepidoptera (moths) and Coleoptera (beetles) at Devilbend Reserve and Mount
676 Worth State Park. Sampling effort = sites × (nights of trapping - light trap malfunctions). Results by vegetation type do not include specimens not
677 identified to a species or morphospecies

Vegetation	2014-2015				2015-2016					
	Sampling effort	Moths		Beetles		Sampling effort	Moths		Beetles	
		Diversity	Abundance	Diversity	Abundance		Diversity	Abundance	Diversity	Abundance
Mount Worth State Park										
Pasture	5	45	282	3	20	6	26	133	5	103
Planting	12	81	910	11	624	12	86	894	19	3,175
Remnant	11	77	726	7	274	12	89	1,057	17	1,057
Total	28	95	2,048	12	1,320	30	102	2,582	21	10,330
Devilbend Reserve										
Pasture	2	19	591	2	2	6	15	178	2	2
Planting	4	44	1,506	2	2	11	46	479	3	4
Remnant	4	56	258	6	95	12	62	327	8	156
Total	10	76	2,477	6	200	29	76	1,140	8	4,791

678

679 **Table 3.** Lepidoptera and Coleoptera with the highest percentage contribution to dissimilarities between vegetation types. Percent contribution
680 was determined using SIMPER analysis of the Bray-Curtis compositional dissimilarity matrix. Key to abbreviations of family names of
681 Lepidoptera: ARCT = Arctiidae, CRAM = Crambidae, EREB = Erebidae, GEOM = Geometridae, HEPI = Hepialidae, LASI = Lasiocampidae,
682 NOCT = Noctuidae, OECO = Oecophoridae and Coleoptera: SCAR = Scarabaeidae

Comparison	Lepidoptera (moths)		Coleoptera (beetles)	
	Species and family abbreviation	Contribution	Species and family abbreviation	Contribution
Pasture vs planting	<i>Hednota pleniferellus</i> ¹ CRAM	12.9%	<i>Telura vitticollis</i> ² SCAR	28.5%
	<i>Proteuxoa sanguinipuncta</i> ¹ NOCT	10.4%	<i>Sericesthis nigrolineata</i> (sp. 33) ^{2,3} SCAR	11.5%
	<i>Elhamma australasiae</i> ² HEPI	8.3%	<i>Scitala sericans</i> ^{2,3} SCAR	8.7%
	<i>Persectania ewingii</i> ¹ NOCT	8.1%	<i>Sericesthis nigrolineata</i> (sp. 42) ^{2,3} SCAR	7.9%
	<i>Abantiades labyrinthicus</i> ⁴ HEPI	7.5%	<i>Phyllotocus nigripennis</i> ^{2,3} SCAR	6.6%
	<i>Persectania dyscrita</i> ¹ NOCT	6.9%	<i>Sericesthis geminata</i> ^{2,3} SCAR	5.9%
	<i>Proteuxoa rubripuncta</i> ¹ NOCT	5.3%	<i>Sericesthis nigrolineata</i> (sp. 41) ^{2,3} SCAR	5.3%
	<i>Hednota grammellus</i> ¹ CRAM	4.7%		
	<i>Pararguda nasuta</i> ⁵ LASI	4.6%		
	<i>Praxis porphyretica</i> ⁵ EREB	3.4%		
Overall dissimilarity		75.6%		82.8%
Pasture vs remnant	<i>Hednota pleniferellus</i> ¹ CRAM	13.6%	<i>Telura vitticollis</i> ² SCAR	24.6%
	<i>Proteuxoa sanguinipuncta</i> ¹ NOCT	9.3%	<i>Sericesthis nigrolineata</i> (sp. 33) ^{2,3} SCAR	10.9%
	<i>Elhamma australasiae</i> ² HEPI	7.0%	<i>Scitala sericans</i> ^{2,3} SCAR	8.3%

	<i>Persectania ewingii</i> ¹ NOCT	6.6%	<i>Sericesthis nigrolineata</i> (sp. 42) ^{2,3} SCAR	8.0%
	<i>Persectania dyscrita</i> ¹ NOCT	6.3%	<i>Sericesthis geminata</i> ^{2,3} SCAR	6.9%
	<i>Pararguda nasuta</i> ⁵ LASI	5.4%	<i>Sericesthis nigrolineata</i> (sp. 41) ^{2,3} SCAR	5.7%
	<i>Abantiades labyrinthicus</i> ⁴ HEPI	5.2%	<i>Phyllotocus nigripennis</i> ^{2,3} SCAR	4.8%
	<i>Palaeosia</i> sp. 2 ⁶ EREB	4.8%	<i>Heteronyx grandis</i> ^{2,3} SCAR	4.3%
	<i>Chlorocoma dichloraria</i> ⁷ GEOM	3.8%		
	<i>Spilosoma (Ardices) canescens</i> ⁸ ARCT	3.8%		
	<i>Garrha</i> sp. 1 ⁹ OECO	3.8%		
	<i>Hednota grammellus</i> ¹ CRAM	3.4%		
Overall dissimilarity		79.8%		79.5%
Planting vs remnant	<i>Proteuxoa sanguinipuncta</i> ¹ NOCT	9.6%	<i>Telura vitticollis</i> ² SCAR	20.2%
	<i>Hednota pleniferellus</i> ¹ CRAM	9.3%	<i>Sericesthis nigrolineata</i> (sp. 33) ^{2,3} SCAR	10.4%
	<i>Abantiades labyrinthicus</i> ⁴ HEPI	6.4%	<i>Scitala sericans</i> ^{2,3} SCAR	8.8%
	<i>Persectania ewingii</i> ¹ NOCT	6.1%	<i>Sericesthis nigrolineata</i> (sp. 42) ^{2,3} SCAR	8.7%
	<i>Elhamma australasiae</i> ² HEPI	6.0%	<i>Sericesthis nigrolineata</i> (sp. 41) ^{2,3} SCAR	6.8%
	<i>Proteuxoa rubripuncta</i> ¹ NOCT	5.1%	<i>Sericesthis geminata</i> ^{2,3} SCAR	6.7%
	<i>Pararguda nasuta</i> ⁵ LASI	5.0%	<i>Heteronyx grandis</i> ^{2,3} SCAR	5.2%
	<i>Persectania dyscrita</i> ¹ NOCT	4.9%	<i>Melolonthinae</i> sp. 17 ^{2,3} SCAR	5.0%
	<i>Palaeosia</i> sp. 2 ⁶ EREB	4.2%		
	<i>Rhapsa suscitatalis</i> ¹⁰ EREB	4.1%		
	<i>Praxis porphyretica</i> ⁵ EREB	4.0%		
	<i>Hednota grammellus</i> ¹ CRAM	4.0%		
	<i>Chlorochoma dichloraria</i> ⁷ GEOM	3.6%		

Overall dissimilarity

74.2%

71.6%

- 683 ¹ Grass (blades) feeding as larvae.
- 684 ² Grass (roots) feeding as larvae
- 685 ³ Eucalypt (leaves) feeding as adults.
- 686 ⁴ Mixed, subterranean (roots) feeding as larvae.
- 687 ⁵ Acacia (leaves) feeding as larvae.
- 688 ⁶ Lichen-feeding as larvae.
- 689 ⁷ Eucalypt (leaves) feeding as larvae.
- 690 ⁸ Mixed, mostly herbaceous dicotyledons (leaves) feeding as larvae.
- 691 ⁹ Fallen leaves (eucalypt) feeding as larvae.
- 692 ¹⁰ No host records (probably not grass-feeding as larvae).

693 **Table 4.** Results of multivariate GLMs on the effect of proximity to remnant vegetation on the abundance of species at a site. Shown here are the
694 coefficient and standard error of models from the mvabund object, the deviance and *P*-value obtained from likelihood ratio tests for the 11 species
695 of Lepidoptera and four species of Coleoptera exhibiting the greatest dissimilarity. Key to abbreviations of family names and host plant associations
696 (denoted by superscripted numbers) as given in Table 3

Lepidoptera (moths)				Coleoptera (beetles)			
Species and family abbreviation	<i>P</i>	Coefficient	Deviance	Species and family abbreviation	<i>P</i>	Coefficient	Deviance
Negative association							
<i>Garrha</i> sp. 1 ⁹ OEEO	0.001	-0.016	14.50	Melolonthinae sp. 35 ^{2,3} SCAR	0.013	-0.032	8.030
<i>Spilosoma (Ardices) canescens</i> ⁸ ARCT	0.021	-0.016	5.081	<i>Webbella firma/Heteronyx</i> sp. ^{2,3} SCAR	0.021	-0.024	5.156
<i>Rhapha suscitatalis</i> ¹⁰ EREB	0.022	-0.027	6.251	<i>Telura vitticollis</i> ² SCAR	0.050	-0.014	3.012
<i>Praxis porphyretica</i> ⁵ EREB	0.025	-0.018	6.191				
<i>Heliomystis electrica</i> ⁷ GEOM	0.025	-0.017	4.314				
<i>Diarsia intermixta</i> ⁸ NOCT	0.038	-0.013	4.791				
<i>Pararguda nasuta</i> ⁵ LASI	0.040	-0.009	4.615				
Positive association							
<i>Elhamma australasiae</i> ² HEPI	0.004	0.013	12.78				
<i>Persectania dyscrita</i> ¹ NOCT	0.006	0.007	8.343				
<i>Persectania ewingii</i> ¹ NOCT	0.011	0.014	9.365				

*Hednota pleniferellus*¹ CRAM

0.016

0.007

6.284

697

698 **Table 5.** Forewing length of Lepidoptera the abundance of which was associated with
 699 proximity to remnant vegetation (see Table 4). Key to abbreviations of family names and host
 700 plant associations (denoted by superscripted numbers) as given in Table 3

Species and family abbreviation	Sex	<i>n</i>	Mean length (mm)	SE
Negative association				
<i>Garrha</i> sp. 1 ⁹ OEEO	-	5	11.2	0.57
<i>Spilosoma (Ardices) canescens</i> ⁸ ARCT	-	5	19.7	0.25
<i>Rhaphsa suscitatalis</i> ¹⁰ EREB	-	5	19.5	0.32
<i>Praxis porphyretica</i> ⁵ EREB	-	5	17.5	0.19
<i>Heliomystis electrica</i> ⁷ GEOM	-	5	20.9	0.08
<i>Diarsia intermixta</i> ⁸ NOCT	-	5	15.2	0.46
<i>Pararguda nasuta</i> ⁵ LASI	-	5	14.4	0.55
Positive association				
<i>Elhamma australasiae</i> ² HEPI	♂	5	17.8	0.49
<i>Elhamma australasiae</i> ² HEPI	♀	5	29.6	0.56
<i>Persectania dyscrita</i> ¹ NOCT	-	5	15.7	0.41
<i>Persectania ewingii</i> ¹ NOCT	-	5	17.3	0.42
<i>Hednota pleniferellus</i> ¹ CRAM	-	5	11.6	0.53

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