1 **Routes to achieving sustainable intensification in simulated dairy farms - the importance of**

2 **production efficiency and complimentary land uses**

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ABSTRACT

1. Sustainable intensification (SI) is a global challenge, aiming to increase food production whilst conserving biodiversity and ecosystem services. This is contrary to the observed trend of agricultural intensification degrading environmental quality. We developed a framework integrating animal nutrition, crop yields, and biodiversity modelling to explore SI potential in multiple model dairy farming systems through varying crop composition to provide cattle feed rations. We then identified key drivers of biodiversity gain that may be applicable at a wider scale.

2. We developed multiple feed rations to meet the nutritional demands of a high-yielding, housed dairy herd. The land area required varied due to productivity and nutritional differences between crops, generating spare land. We used published biodiversity models to compare alpha- and beta-diversity of spiders and plants across 36 scenarios that used the spare land in different ways, for either biodiversity maximisation or additional production.

3. Alpha and beta-diversity for both taxa was greatest in scenarios that maximised spare land and utilised this for species-rich extensive grassland. However, commensurate biodiversity gains for plant alpha-diversity, and spider and plant beta-diversity (respectively 100%, 76% and 86% gain relative to that optimal scenario) were achievable when spare land was used for additional crop production.

4. Maximising compositional heterogeneity and adding complementary productive land uses to spared land were key to increasing production and beta-diversity, while adding species-rich productive land uses drove increasing production and alpha-diversity.

5. *Synthesis and applications.* This study indicates the potential for SI of dairy farming through manipulating feed rations to increase land-efficiency and spare land, which could then be used to enhance production and biodiversity. The optimum land composition depends on target goal(s) (e.g. maximising production and/or biodiversity). Greatest 'win-wins' were

45   achieved through increasing land cover heterogeneity and selecting crops that complement

46   each other in the species they support, highlighting the important role of heterogeneity in

47   the crop matrix. Our study provides a framework that integrates production efficiency and

48   biodiversity modelling to explore potential routes to achieve SI goals.

49

50   **KEYWORDS**

51   agriculture, alpha-diversity, beta-diversity, biodiversity, dairy, heterogeneity, agricultural production,

52   sustainable intensification

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INTRODUCTION

The challenge of protecting biodiversity and ecosystem services in the face of increasing food

production has led to the call for sustainable intensification (SI) of agricultural land; that is, the

simultaneous increase of food production and reduction of environmental pressure (Tilman *et al.*

2011; Garnett *et al.* 2013). This is now a key strategic objective of the UN's Food and Agriculture

Organisation (FAO 2015). A premise of SI is that increased production should be achieved on existing

farmland, because clearance of other habitats for agriculture creates greater environmental

degradation and biodiversity loss (Tilman *et al.* 2011; Garnett *et al.* 2013). Thus, a question central to

SI is whether we can increase production of existing land while maintaining or improving its

biodiversity.

Livestock production and biodiversity

Demand for meat and dairy products have increased substantially, with global cattle stock increasing

by 57% between 1961 and 2016 (FAOSTAT 2017). Grassland management has been intensified,

changing the functional composition of vegetation, with associated global biodiversity declines being

observed across taxa (Plantureux, Peeters & McCracken 2005). Functional and/or taxonomic

changes to vegetation composition have benefitted some ecosystem services (e.g. food provisioning

/ nutrient cycling services), at the cost of others (e.g. regulating, cultural and biodiversity services)

(Allan *et al.* 2015).

To satisfy the nutritional demands (i.e. fat, carbohydrate, protein) of high-yielding dairy cows ($\geq$40 L

milk d^{-1} animal^{-1}), feed rations comprise of multiple crops (Toma *et al.* 2013). Variation in the

composition of animal feeds and in-crop productivity produces large differences in the amount and

composition of land required to fulfil the dietary needs of a given herd at a target milk yield. In the

context of SI, this creates a dual opportunity: (1) if land-use composition can be modified to increase

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80   production, modifications may be steered towards productive land compositions that maintain, or

81   even improve, biodiversity; (2) if efficiency improvements mean land-use composition can be

82   modified to produce spare land, whilst still maintaining or increasing production, this land might be

83   used for high biodiversity land-uses. Grassland simulations indicate heterogeneity of management

84   intensity can be varied to simultaneously increase arthropod populations and food production

85   (Simons & Weisser 2017). However, this has not been explored in mixed land-use systems where

86   there may be greater scope to manipulate crop composition to achieve multiple benefits.

87

88   Our study integrates animal nutrition, crop productivity, and biodiversity modelling to explore a

89   range of land-use scenarios that meet the feed requirements of a model dairy system to determine

90   the possibility of increasing food production whilst maintaining or enhancing biodiversity. We

91   developed a range of scenarios and manipulated the composition of land-covers to determine where

92   'win wins' could be achieved and to identify key drivers of production and/or biodiversity gains. All

93   scenarios provided feed for a herd of at least 100 permanently housed dairy cattle but differed in

94   terms of their land-use composition and thus land-efficiency (Fig. 1). More land-efficient scenarios

95   generated up to 18% 'spare land', which could then be used for additional production, or allocated

96   to a biodiversity-rich habitat (i.e. extensive grassland). Extending published land-use/biodiversity

97   models, we estimated indices for scenarios' alpha and beta-diversity for two functional groups

98   (plants and epigeal spiders) where extensive biodiversity data were available (Downie *et al.* 1999;

99   Wilson *et al.* 2003). We hypothesised that production of the system could be maintained or even

100   improved, whilst also improving or maintaining diversity of plants and spiders. We thus explore the

101   potential for SI within a dairy system and discuss implications for achieving SI goals more widely.

102

103   **MATERIALS AND METHODS**

104   **Land-use scenarios**

105   Our modelling process explored a 'home-grown feeds' dairy system, where nutritional requirements

106   of permanently-housed dairy cows are grown on the farm (Roberts & March 2014). We used an

107   industry-standard livestock nutrition feeding model, FeedByte (Scottish Agricultural College 2006), to

108   design alternative feed rations for a model herd of 100 Holstein-Friesian cows with a target milk

109   yield of 9,500 L yr^{-1} (equivalent to 40 L of milk cow^{-1} day^{-1} including an unproductive period). This

110   commercially-used software uses least-cost diet formulation and linear-programming and is used by

111   industry and researchers to design real feed rations sufficient for livestock herds (Chagunda,

112   Flockhart & Roberts 2010). Rations needed to comprise of grass silage, barley, wheat, oilseed rape

113   and fodder beet as comprehensive biodiversity data existed for these crops and they varied with

114   respect to the availability of specific nutrients. Each crop alone would not provide the combination

115   of nutrients (e.g. proteins, carbohydrates or fats) required for high-yielding cattle. For each ration,

116   we used productivity estimates using both fresh-weight and dry-matter (DM) yields (kg ha^{-1}) as well

117   as nutritive value estimates (g kg^{-1}) for the constituting crops (Supporting Information Table S1) to

118   calculate the land area required. Due to differences in crop yields and nutritive values, the amount

119   of land required to meet the herds' nutritional requirements differed substantially between rations.

120   The feed ration requiring the most land for our target milk yield and herd size (i.e. the least land-

121   efficient ration) was designated the 'baseline scenario' (103.8 ha) (Fig. 1a). Our most land-efficient

122   feed ration provided the same milk yield and herd size on less land (88.0 ha). Consequently through

123   manipulating the quantities of different crops in the feed ration we could generate up to 15.8 ha

124   (18%) of spare land relative to the baseline. This generated our maximum spare land test-system

125   (Fig. 1b).

126

127   To compare a variety of scenarios where feed rations are met via different land areas and

128   compositions, we generated five intermediate test-systems varying in land-efficiency between the

129   baseline scenario and the max. spare land test-system (1%, 6%, 8%, 12% and 15% spare land c.f. the

130   baseline). The result was six test-systems (Fig. 1b) containing spare land; the choice of six allowed

131   generation of 36 'spare-land scenarios', considered a suitable sample size for exploring drivers of

132   biodiversity change across scenarios. We generated spare-land scenarios by replacing the spare land

133   component of each test-system (Fig. 1b) with one of six land uses (grass silage, barley, wheat,

134   oilseed rape, fodder beet or extensive grassland) (Fig. 1d). Spare land scenarios thus differed in both

135   the amount of spare land and the land use replacing the spare land component. Spare land scenarios

136   could be divided into:

137

138   (1) 'additional production' scenarios where spare land was used to grow more of one of the

139   productive crops (silage, wheat, barley, oilseed rape or fodder beet; Fig. 1d i).

140

141   (2) 'no additional production' scenarios where spare land was designated to extensive grassland (Fig.

142   1d ii). Of the land uses available in our empirical datasets, we selected extensive grassland to

143   represent a high-biodiversity, low/non-productive land use with no improvement or cutting regime.

144

145   Sparing land from production and designating to biodiversity-rich habitats, as in our 'no additional

146   production' scenarios, is a frequent focus of agri-environment prescriptions (e.g. fallow, floristically

147   diverse field margins). However, the original datasets we used (see below) did not contain data from

148   such habitats so we used data from extensively grazed, semi-natural grassland (e.g. calcifugous and

149   *Juncus*-dominated) to represent a low-production, high biodiversity land use as our alternative

150   strategy to increasing production. These are not directly equivalent to newly created agri-

151   environment habitats, since environmental (e.g. topography, altitude, soil, climate) and socio-

152   ecological (grazing regimes and underlying productivity of the land) constraints would have

153   historically prevented intensification on such areas with long-term extensive management resulting

154   in unique, species-rich, communities (Downie *et al.* 1999; Wilson *et al.* 2003; Cole *et al.* 2005). We

155   discuss the implications of this for our findings below.

156

157   We also explored an additional scenario where the desired outcome was to maximise milk

158   production ('max. herd size': Fig. 1c). This was achieved by scaling up the land composition of the

159   most land-efficient test-system, which had 18% spare land ('max. spare land'; Fig. 1b), to occupy the

160   area of the baseline scenario resulting in an increase from 100 to 118 cattle and thus increasing milk

161   production by 18%. Thus we generated 38 scenarios in total: the baseline scenario (Fig. 1a), the

162   max. herd size scenario (Fig. 1c), and the 36 spare land scenarios (Fig. 1d).

163

164   **Alpha diversity, beta-diversity, and additional production estimates**

165   For each of the 36 spare land scenarios (Fig. 1d) and the max. herd size scenario (Fig. 1c), we

166   calculated indices of alpha and beta-diversity for plants and spiders, and compared these with the

167   baseline scenario. This analysis framework is illustrated in Fig. 2. Diversity indices were generated

168   from published biodiversity models derived from field studies measuring plant and spider species

169   richness in the agricultural land-covers in our system (Downie *et al.* 1999; Wilson *et al.* 2003).

170

171   Data had been collected in nine geographical locations in Scotland ranging from intensive arable and

172   grassland landscapes, mixed farming and crofting. Vegetation surveys were conducted over three

173   years (1995-1997) at 87 sites with data being collected from permanent 10×10 m quadrats on three

174   occasions during peak growing season (June-September) (Wilson *et al.* 2003). Epigeal spiders were

175   surveyed over two years (1996-1997) at a total of 71 sites using a 16 m transect of nine pitfall traps

176   (75 mm diameter and 100 mm deep). Pitfall trapping was conducted May-September with contents

177   collected monthly (Downie *et al.* 1999).

178

179   Plants and spiders are key components of agricultural ecosystems, strongly driven by land

180   management (Wilson *et al.* 2003; Batáry *et al.* 2012). Plants are key drivers of invertebrate

181   biodiversity (Symstad, Siemann & Haarstad 2000) and provide shelter and breeding sites for many

182   species, while spiders are important polyphagous predators contributing to natural pest control

183   (Herzog *et al.* 2013). Alongside wild bees and earthworms, they provide appropriate and robust

184   metrics for measuring agricultural biodiversity (Herzog *et al.* 2013).

185

186   To generate alpha-diversity estimates for spiders and plants we used species richness (S) models

187   fitted by Downie *et al.* (1999) and Wilson *et al.* (2003). These models were developed to explore

188   drivers of field-scale S in a range of land uses. The process we used to generate an alpha-diversity

189   index (ADI) for each scenario is described in detail in Supporting Information SECTION B. Briefly, the

190   ADI for each scenario is defined as the mean predicted field-scale S across land-uses, weighted by

191   the proportional area of each land-use. Weighted mean S is a recognised metric to explore impacts

192   of agricultural land-use composition on biodiversity (e.g. Hiron *et al.* 2015).

193

194   To generate beta-diversity indices we used data describing the assemblage structure of our target

195   groups collected by Downie *et al.* (1999) and Wilson *et al.* (2003). Beta-diversity here describes the

196   extent of variation of species assemblages between different land-uses. Raw species data for the

197   models considered in the ADI analyses were not available. However, detrended correspondence

198   analysis (DCA) axes scores derived from the raw data were available (Murphy *et al.* 1998). The

199   distance between two sites in a DCA ordination provides a measure of similarity in species

200   composition, with smaller distances indicating greater similarity (Smol *et al.* 2005). The process we

201   used to generate beta-diversity indices (BDI) for each scenario is described in detail in Supporting

202   Information SECTION B. Briefly, the BDI for each scenario is defined as the median DCA distance

203   between random pairs of land-uses within each scenario, weighted by the proportional area of each

204   land-use. We provide R code for the estimation of ADI and BDI in Supporting Information SECTION C

205

206   While each spare-land scenario (Fig. 1d) maintained our target milk yield, a scenario's 'value' can be

207   modified relative to the baseline scenario in two main ways: (i) biodiversity value, indicated by ADI

208   and BDI; and (ii) production value, which can be increased when the spare-land component is used

209   for additional production. To determine how spider and plant ADI and BDI changed relative to the

210   baseline scenario we subtracted the baseline ADI and BDI values from those calculated for each

211   scenario, the results termed Δ_{ADI} and Δ_{BDI} (Fig. 2).

212

213   To allow us to simultaneously explore both the change in biodiversity and production for additional

214   production scenarios, we estimated the production capacity of the spare land component as

215   metabolisable energy ('ME', MJ kg^{-1} DM) to standardise this benefit across different crops. ME

216   provided a more direct measure of production capacity than monetary value of crops, which is

217   dependent on a range of additional market factors. We derived ME of the spare land component for

218   each additional production scenario by multiplying industry estimates of ME by yield (kg DM ha^{-1})

219   (SAC Consulting 2014) and area (ha). The baseline scenario did not produce any additional energy on

220   top of feeding the herd of 100 cattle, and consequently the calculated ME value reflects change

221   relative to the baseline (i.e. Δ_{ME}).

222

223   **Drivers of relative alpha and beta-diversity**

224   To investigate which factors drove the magnitude of Δ_{ADI} or Δ_{BDI} for each taxa, we generated four

225   candidate sets of general linear mixed models, with Δ_{ADI} or Δ_{BDI} for plants and spiders as the

226   response variables. Each spare land scenario provided a single estimate of Δ_{ADI} and Δ_{BDI} (Fig. 2) so the

227   sample size for the analysis was 36. We included test-system identity (Fig. 1b), and land-use

10

228   constituting the spare-land component (Fig. 1d) as random effects. We describe the fixed effects

229   included in our models in Table 1.

230

231   For each taxa (plant/spider) and response variable (Δ_{ADI}/Δ_{BDI}) we compared models using a small-

232   sample Akaike's Information Criterion (AICc) and Akaike's weights (w_i) (Johnson & Omland 2004).

233   We included several multivariate models and the appropriate null model. We restricted inference on

234   the key drivers of Δ_{ADI} or Δ_{BDI} to examining the smallest set of models containing a summed $w_i \geq 0.90$.

235

236   **RESULTS**

237   **Impact of land use on alpha and beta diversity**

238   Predicted estimates of S for spiders and plants from the alpha-diversity models for individual land-

239   uses are shown in Figs. 3a-b. Predictions of spider S were as low as 11.7 (lower 95% confidence limit

240   [CL_{95}] for silage) and as high as 41.9 (upper CL_{95} for extensive grassland). For plants, values ranged

241   from 6.1 species (lower CL_{95} for barley) to 29.6 (upper CL_{95} for extensive grassland). These were

242   within the observed field-scale ranges from the original data-sets of 10-56 spider species (Downie *et*

243   *al.* 1999) and 5-57 plant species (Wilson *et al.* 2003).

244

245   Assemblage structure of spider and plant communities was most similar in oilseed rape and cereals

246   (i.e. barley and wheat) with communities in these crops showing the greatest disparity with those in

247   extensive grasslands and fodder beet (Figs. 3c-d).

248

249   **Comparison of alpha and beta diversity between scenarios**

250   The baseline scenario had a predicted ADI (the mean field-scale S across land-uses, weighted by the

251   proportional area of each land-use) of 19.8 (CL_{95} 17.8-22.8) spider species and 13.4 (CL_{95} 11.9-15.2)

252   plant species. Δ_{ADI} and Δ_{BDI} for each taxa and scenario are shown in Fig. 4. In all cases, 'no additional

253 production' scenarios with maximum spare land gave the highest biodiversity benefit (labelled 'NAP'

254 in Fig. 4). However, for both taxa Δ_{BDI}, and for plant Δ_{ADI}, additional production scenarios existed that

255 generated similar biodiversity gains to the latter (optimal scenarios labelled 'AP' in Fig. 4). For

256 spiders, additional production scenarios did not enhance ADI due to the baseline scenario having a

257 relatively high ADI (Fig. 4a).

258

259 **Drivers of change in alpha and beta-diversity**

260 For both spider and plant Δ_{ADI}, a single model was in the confidence set (Table 2). This contained the

261 estimated S of the added land-use, the area of spare land, and their interaction (Figs. 5a-b). Adding a

262 land-use with a comparatively high estimated S to the spare-land component increased Δ_{ADI}, and this

263 interacted positively with amount of spare land.

264

265 For spider Δ_{BDI}, two models were in the confidence set (Table 2), containing the mean DCA distance

266 for the land-use being added, the land-use evenness ($w_i = 0.87$) or heterogeneity ($w_i = 0.08$), and

267 their interaction. A higher mean DCA distance of the land-use being added increased Δ_{BDI}, but only

268 where land-use evenness was also increased (Fig. 5c). For plant Δ_{BDI}, four models were in the

269 confidence set (Table 2). The two best models contained either the land-use heterogeneity ($w_i =$

270 0.71) or evenness ($w_i = 0.13$), showing positive relationships (Fig. 5d).

271

272 **'Value' of spare-land scenarios**

273 For both Δ_{ADI} (Fig. 6a) and Δ_{BDI} (Fig. 6b) utilising spare land as extensive grassland delivered the

274 optimal biodiversity gain across taxa, but generated no Δ_{ME} gain. For Δ_{BDI}, not only was a max. spare

275 land + fodder beet scenario able to achieve gains almost commensurate with the best no additional

276 production scenario for both spiders and plants, but it did so while achieving the highest Δ_{ME} (Fig.

277 6b).

278

279 For Δ_{ADI}, the relatively high ADI value for spiders in the baseline scenario resulted in a complex

280 picture (Fig. 6a). All additional production scenarios showed a reduced ADI for spiders, but increased

281 ADI for plants. Some added fodder beet scenarios achieved relatively high plant Δ_{ADI} and moderate

282 Δ_{ME} gains but at a cost to spider Δ_{ADI} values (albeit with spider losses lower in magnitude than plant

283 gains). Some added silage scenarios achieved greater plant Δ_{ADI} gains than added fodder beet (equal

284 to plant Δ_{ADI} gain in the best no additional production scenario) but at a cost to spider Δ_{ADI} and a

285 lower Δ_{ME}, indicating trade-offs between potential gains.

286

287 **DISCUSSION**

288 **Sustainable intensification outcomes**

289 Simultaneous gains in production and biodiversity were simulated in our model dairy system

290 illustrating the potential for SI. We achieved the biggest simultaneous gains in beta-diversity and

291 production by increasing the system's efficiency to feed the dairy herd using a smaller area of land,

292 and allocating the spare land generated to a productive crop that complemented those already

293 present in the system with respect to the species supported. For plant alpha-diversity, we achieved

294 highest gains in production and species richness by allocating the spare land to a species-rich,

295 productive land use.

296

297 Our simulations challenge the well-observed negative relationship between productivity and

298 biodiversity in agriculture (Krebs *et al.* 1999). SI has been demonstrated in small farms in developing

299 countries using a combination of approaches including integrated plant nutrient systems, no-

300 till/conservation agriculture and integrated pest management (Chappell & LaValle 2011). Evidence of

301 SI in highly-productive agricultural systems is less well documented with gains in biodiversity

302 typically being accompanied by yield losses (Gabriel *et al.* 2010; Firbank *et al.* 2013). SI appears

13

303 particularly challenging in intensive dairy farms - a study exploring innovative management practices

304 found dairy farms notable in their failure to achieve simultaneous production and biodiversity gains

305 (Firbank *et al.* 2013). Our results highlight a potential route to SI in home-grown dairy systems.

306

307 With increasing pressure on our finite agricultural land, SI is important both for local and wider scale

308 biodiversity outcomes. Our 'no additional production' scenarios, increased farm-scale biodiversity,

309 but without increasing production. Increasing demand for food would therefore have to be met by

310 intensification of other farmland and/or generating new farmland (see Fischer *et al.* 2014), with

311 potentially negative biodiversity implications at a wider scale. Our 'additional production' scenarios

312 that achieved SI would not only benefit local biodiversity but also reduce production pressure on

313 land elsewhere. Ultimately, SI solutions will be important in allowing biodiversity to be increased

314 both on a farm scale through land-sharing, and by increasing production on existing farmland which

315 could protect biodiversity-rich areas at the landscape scale ('land-sparing'). The land sharing/sparing

316 debate, however, can overlook the wide range of ecosystem services agricultural land can provide

317 (Fischer *et al.* 2014) and measures that promote biodiversity often improve landscape multi-

318 functionality enhancing a wide-range of ecosystem services (Allan *et al.* 2015).

319

320 **Potential for implementation to real systems**

321 A key finding was the important role that crop heterogeneity played in increasing beta-diversity and

322 production. Loss of heterogeneity (of both semi-natural and agricultural habitats) is a key driver of

323 biodiversity declines, and its restoration represents a mitigation strategy (Benton, Vickery & Wilson

324 2003). Agricultural policies aimed at stemming biodiversity loss typically incentivise farmers to

325 increase heterogeneity via agri-environment schemes (AES), which may involve in-production (land-

326 sharing) or out-of-production (land-sparing) approaches (Batáry *et al.* 2015). Restoring semi-natural

327 habitats in intensive agricultural landscapes typically involves removing land from production, to

328   increase spatial and structural heterogeneity (Ovenden, Swash & Smallshire 1998). Our results

329   suggest that incentivising farmers to increase heterogeneity of their productive land could lead to

330   biodiversity gains whilst increasing production. At a regional level, that could involve a coordinated

331   approach to produce crops that have high complementarity and are not already dominant.

332

333   In introducing a crop diversification component within its compulsory greening measures (EU

334   Regulation 1307/2013), the EU's 2014 Common Agricultural Policy reform may represent a step in

335   the right direction. However, 'diversification' in this legislation narrowly focuses on compositional

336   heterogeneity, bringing its effectiveness under question. Josefsson et al. (2017) found that crop

337   composition *per se* did not influence bird species richness, but structural crop heterogeneity did

338   have a positive effect. For such measures to make a real impact, they may need to go further and be

339   backed-up by a political will to diversify farmland. Some AES are piloting a cluster farming approach

340   to provide landscape-scale benefits (e.g. Natural England 2017); SI strategies could also potentially

341   operate at a multi-farm scale.

342

343   With SI goals aiming to increase production and thus income, appropriate knowledge exchange may

344   enhance uptake of SI-focussed management without additional incentives, although in the UK some

345   innovative farms seen to achieve SI outcomes had relied on AES subsidies to enhance biodiversity

346   (Firbank *et al.* 2013). Improving links between researchers, advisors and farmers are identified as

347   important for uptake of SI practices in developing countries (Pretty, Toulmin & Williams 2011).

348   Promoting biodiversity can have direct economic benefits through enhancing ecosystem services

349   such as natural pest control and pollination, potentially contributing to SI via ecological

350   intensification (Tscharntke *et al.* 2012). Promoting the potential economic benefits of enhancing

351   biodiversity may also be important (Pywell *et al.* 2015). For example, insect pollination can increase

352 yield of oilseed rape (Bommarco, Marini & Vaissière 2012) while the presence of natural enemies of

353 aphids can increase yield of barley (Östman, Ekbom & Bengtsson 2003).

354

355 Incentivising heterogeneity requires careful consideration of costs and benefits. Crop heterogeneity

356 may promote economic sustainability, providing insurance against unpredictable growing conditions,

357 pest or disease outbreaks, or market variability (Garnett *et al.* 2013). In our home-grown system, the

358 production of additional crops not directly required for *in situ* cattle, requires additional market

359 engagement, but markets can be unpredictable. Growing new crops may also incur direct costs, such

360 as those for new machinery or alternative agrochemicals. Our simple measure of increased

361 production was solely based on the energetic yield of a crop, while demand depends on other socio-

362 economic drivers such as available income and consumer behaviour (Valin *et al.* 2014).

363

364 Simultaneously enhancing biodiversity and production clearly requires incorporating complex

365 information derived from disparate sources. For example, here we integrated biodiversity data from

366 both published literature, dietary modelling that combines nutritional equations to determine

367 protein and energy requirements, and finally crop production data. Farmers are familiar with a range

368 of decision-support tools, such as nutrition models, yield estimates, agronomist advice and

369 guidelines on incentives for AES. Comprehensive biodiversity datasets that evaluate biodiversity

370 across habitats are, however, often only available as summary information in scientific publications

371 (e.g. Cole *et al.* 2017). This could make it difficult for farmers to adequately consider biodiversity in

372 decision-making processes. There is great potential for existing monitoring schemes (e.g. UK's

373 Breeding Bird Survey: https://www.bto.org/volunteer-surveys/bbs; UK's Butterfly Monitoring

374 Scheme: http://www.ukbms.org/) to assist in the collection of comprehensive biodiversity data

375 across taxa and land covers, but more fundamental is providing the resultant data to land managers

376 in a usable format. Our framework highlights the potential for current databases (e.g. crop

377   productivity and land-cover/biodiversity) to be integrated to create a SI decision-support tool. In

378   addition to including production and biodiversity outcomes, such a tool could also include other

379   environmental (e.g. reducing greenhouse gas emissions, mitigating diffuse pollution) and agronomic

380   (e.g. reduced agrochemical applications) benefits.

381

382   Simulation studies are important first steps in assessing potential SI outcomes (e.g. Simons &

383   Weisser 2017). Ultimately, however, model predictions require trialling in real landscapes. While

384   simulated studies provide data-driven working hypotheses, they have inevitable limitations. For

385   example, we considered compositional but not configurational heterogeneity, which can drive

386   biodiversity patterns in birds (Hiron *et al.* 2015). In addition, data constraints meant that to

387   represent high-biodiversity, low-input habitats in our 'no additional production' scenarios we used

388   data from historical extensively grazed grasslands rather than potentially more appropriate AES

389   prescriptions (i.e. habitats formed from previously cultivated land that have not had a history of low-

390   input management). Extensive grasslands support unique species assemblages, and it is unlikely that

391   AES prescriptions (e.g. species-rich field margins) would reach the potential biodiversity value of

392   such grasslands (Downie *et al.* 1999; Wilson *et al.* 2003). Our 'no additional production scenarios'

393   may therefore over-estimate biodiversity gains. Agri-environment interventions can, however, not

394   only increase ecological connectivity and provide ecological contrast that enhances resource

395   diversity, stability and availability supporting a wider suite of species (Batáry *et al.* 2015; Cole *et al.*

396   2017), but can also enhance biodiversity-dependant ecosystem services thus benefitting production

397   (Pywell *et al.* 2015). Landscape trials of scenarios could elucidate effects of configurational,

398   structural and compositional heterogeneity, and test biodiversity benefits of AES prescriptions

399   against our assumptions using historical extensively grazed habitats.

400

401   A key reason for investigating potential for SI outcomes in a home-grown dairy system was the

402   closed nature of the system, whereby the land required to feed the herd is *in situ*. Feeding housed

403   cattle is a common dairy production system (e.g. 55% of UK dairy farms house cattle year-round, a

404   percentage which is growing - March *et al.* 2014) but housed cattle can be fed on home-grown or

405   bought-in feeds, or a combination. Home-grown systems represent one end of a spectrum of

406   possible feeding systems for housed dairy cattle (Roberts & March 2014), one which is fully self-

407   sufficient in terms of feeds. While data on the extent of feed self-sufficiency of farms are not widely

408   available, it can be regionally high (e.g. averaging 79-85% in W France: Brocard *et al.* 2016). Higher

409   self-sufficiency of feeds is seen to reduce consumption of non-renewable energy (i.e. reduced

410   transport of bought-in feeds) and raise nutrient efficiency (Gaudino *et al.* 2018). In Europe,

411   producing local sources of protein for dairy cattle is likely to reduce reliance on imported soybean

412   *Glycine max* imports, associated with high greenhouse gas emissions (Hörtenhuber, Lindenthal &

413   Zollitsch 2011), while programs have been established aiming to increase levels of dairy feed self-

414   sufficiency (Ineichen *et al.* 2014). Our simulations demonstrate a further potential environmental

415   benefit of home-grown systems - that composition of feed crops could be managed locally to

416   provide simultaneous productivity and biodiversity gains.

417

Conclusions and application

418

419   Achieving SI is an important but ambitious aim (Tilman *et al.* 2011; Garnett *et al.* 2013). Our study

420   combined biodiversity data, yield data and nutrition models to find routes to simultaneous increases

421   in production and biodiversity in a home-grown dairy system. This was optimised by maximising

422   land-efficiency and targeting additional production to a land cover that had relatively high species

423   richness (alpha-diversity), and complemented species in existing crops (beta-diversity). This

424   highlights the importance of integrating agronomic efficiency, land cover heterogeneity and species

425   richness/complementarity of both productive and non-productive land covers within an SI

426   framework. Where comprehensive biodiversity/production data exist, our framework could be

427   adaptable to other taxonomic groups, production systems and regions. Agri-environment policy

428   should focus not only on increasing the quality and heterogeneity of semi-natural habitats, but also

429   on enhancing agricultural efficiency and the complementarity and heterogeneity of productive land

430   covers. Through developing a framework that integrates crop productivity and biodiversity

431   modelling to seek optimal production-biodiversity scenarios, this study presents a route to identify

432   key drivers of production and biodiversity gain, a key goal of SI, that may be applicable at a wider

433   scale.

434

435   **AUTHORS' CONTRIBUTIONS**

436   All authors developed the initial concepts and MAL designed test scenarios. PJCW conducted

437   analyses and LJC sourced the datasets. PJCW and LJC wrote the initial draft and all authors

438   contributed to revisions.

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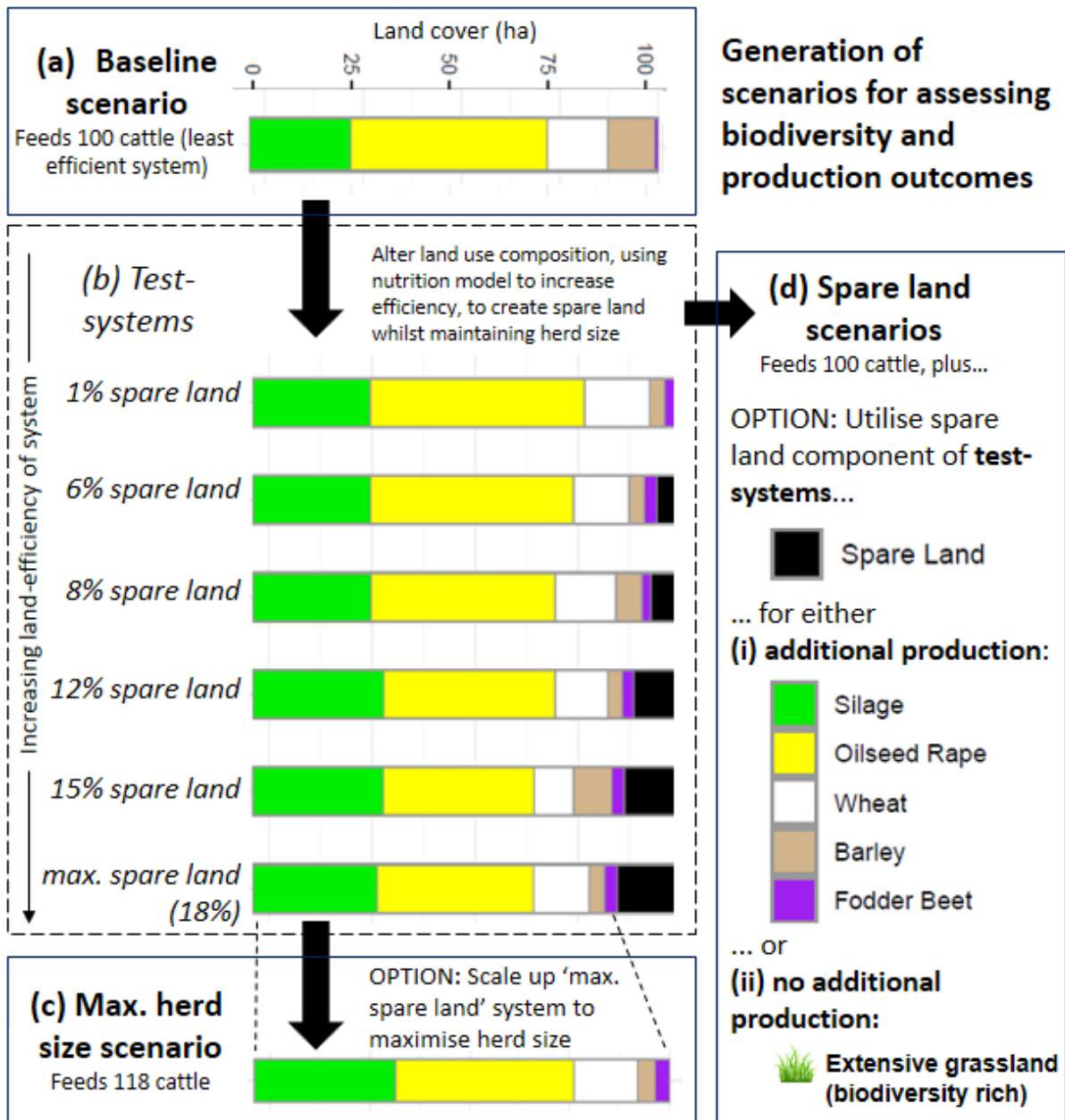
FIGURE 1. The generation of 38 dairy system scenarios. The feed ration requiring the most land

(103.8 ha) was designated the baseline scenario. Altering the farm's crop composition generated

spare land (b), without reducing herd size or milk yield (see text). This spare land could be used to

(c) scale up the system to maximise herd size, (d i) for additional production or (d ii) for

maximising biodiversity through addition of extensive grassland (no additional production).

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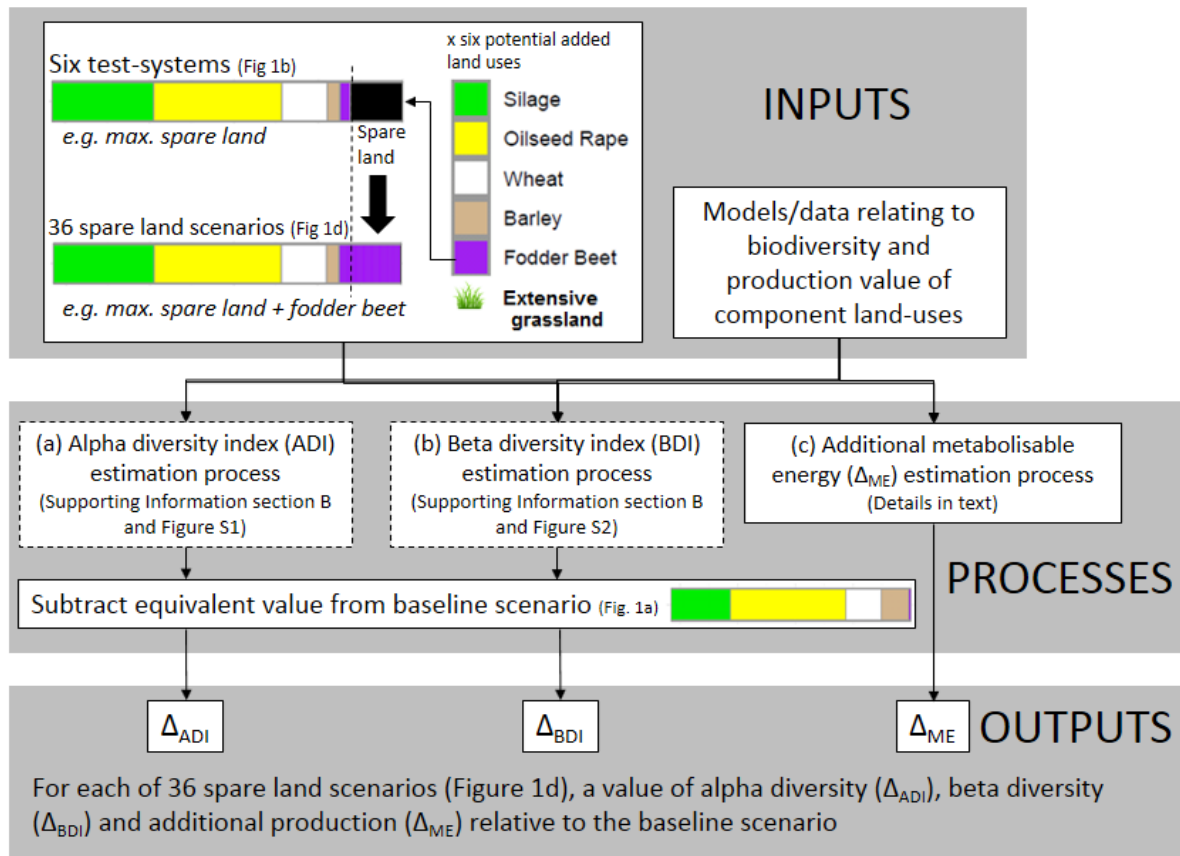
FIGURE 2. Workflow for producing estimates of alpha-diversity, beta-diversity and additional production for the 36 spare land scenarios (Fig. 1d), relative to the baseline scenario (Fig. 1a). Further information is provided in the text. More detailed information on the estimation of (a) alpha-diversity and (b) beta-diversity indices (shown in dashed boxes), with sub-workflows and model code, are provided in the Supporting Information.

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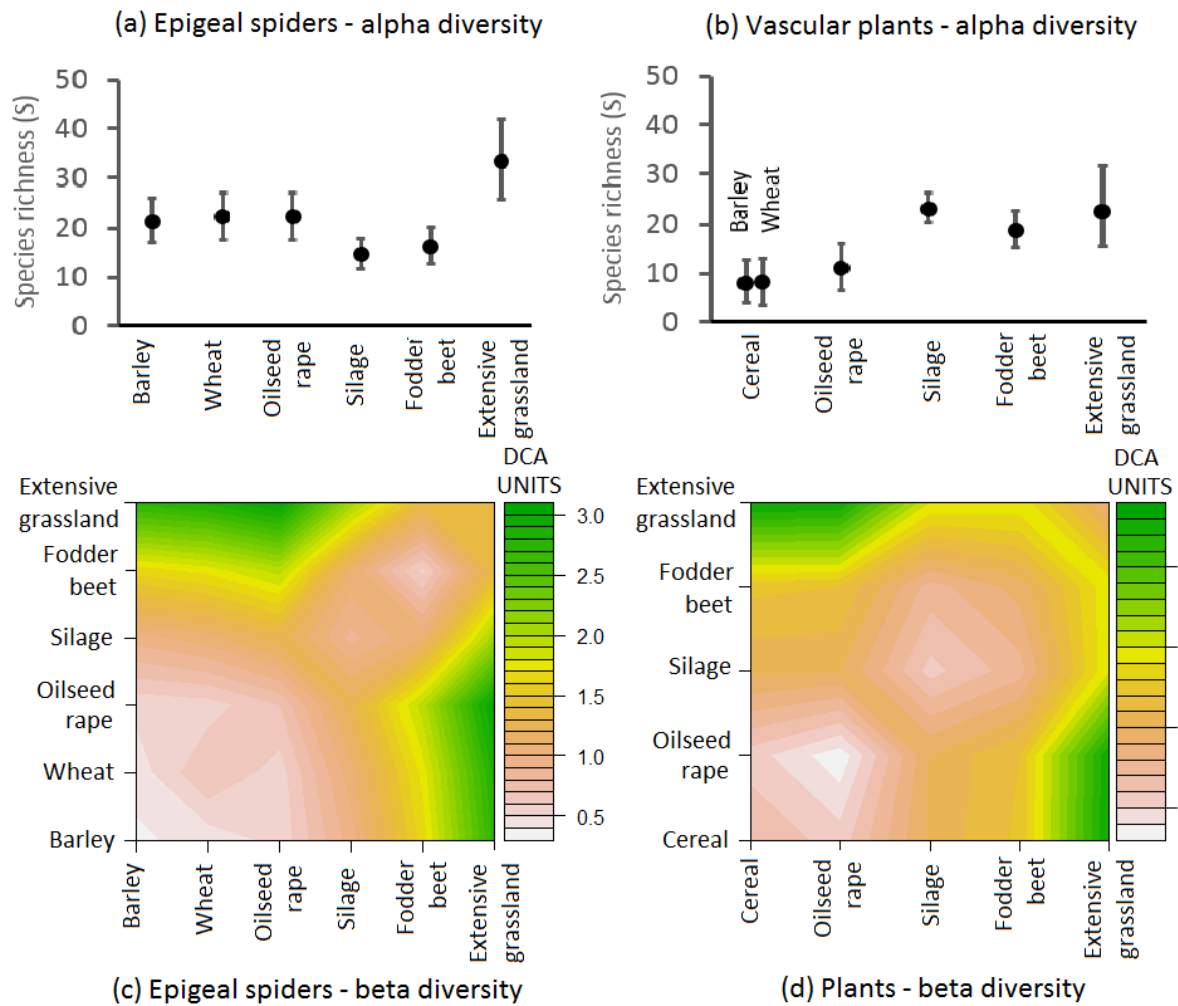
FIGURE 3. Alpha and beta-diversity estimates for plants and epigeal spiders for each land-use from datasets in Downie *et al.* (1999) and Wilson *et al.* (2003): (a)-(b) estimated field-scale species richness (bars representing upper and lower 95% confidence limits); (c)-(d) contour plots representing mean detrended correspondence analysis (DCA) distances between and within land-uses (a measure of beta-diversity). Land-uses are ordered to minimise DCA distances across the primary and secondary diagonals in (c)-(d).
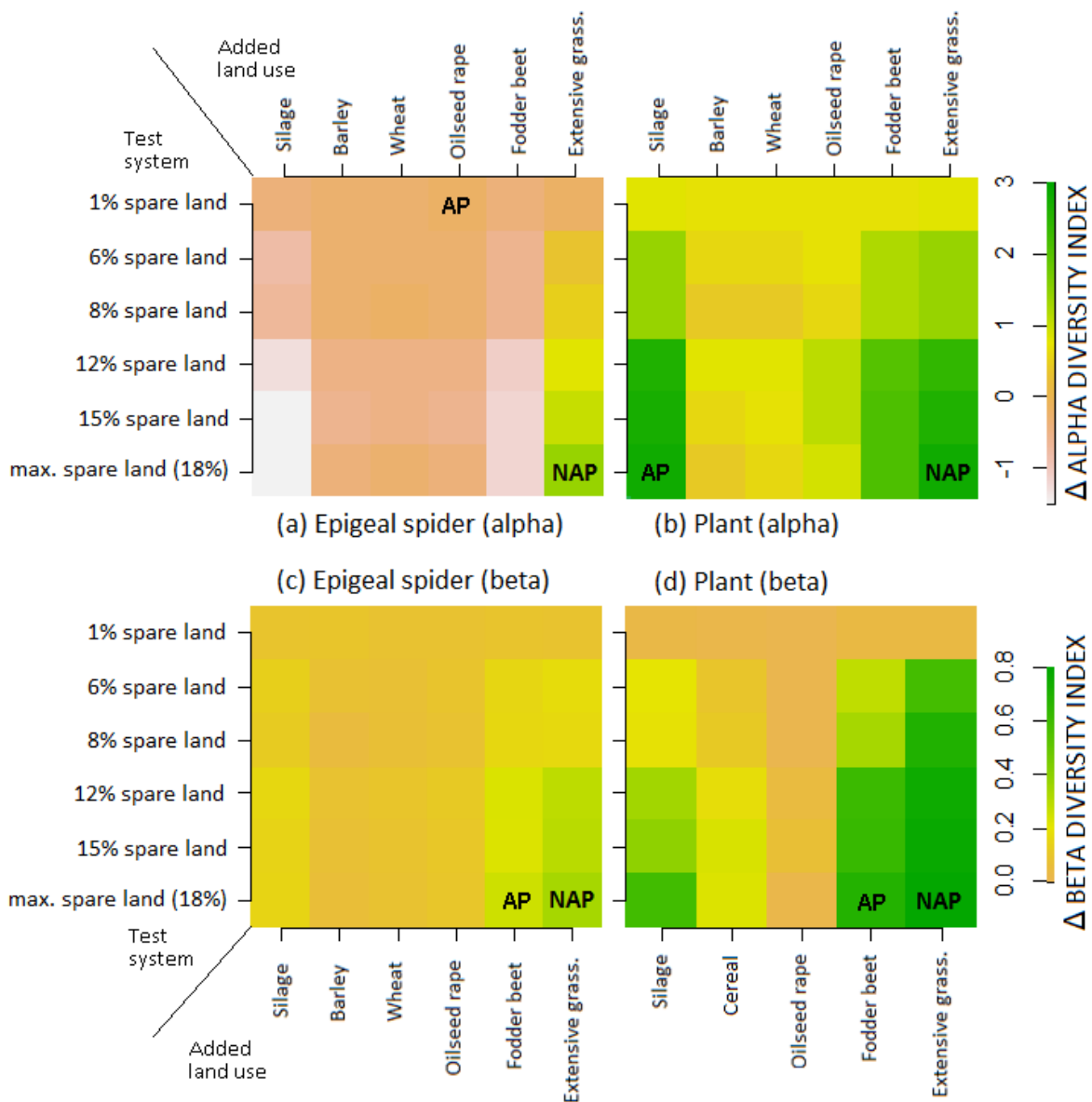
FIGURE 4. The (a)-(b) change in alpha-diversity index (Δ_{ADI}) and (c)-(d) change in beta-diversity index (Δ_{BDI}) for 36 spare-land scenarios relative to the baseline scenario, for spiders and plants. Each spare-land scenario is defined by a test-system (y-axis) and a land-use that is added (x-axis) to the spare land component of that test-system (Fig. 1). For each plot, the scenario marked 'AP' represents the highest value for an 'additional production' scenario, and 'NAP' represents the highest value for a 'no additional production' scenario (Fig. 1d). Derivation of alpha- and beta-diversity indices from field-scale biodiversity data are described in the text.

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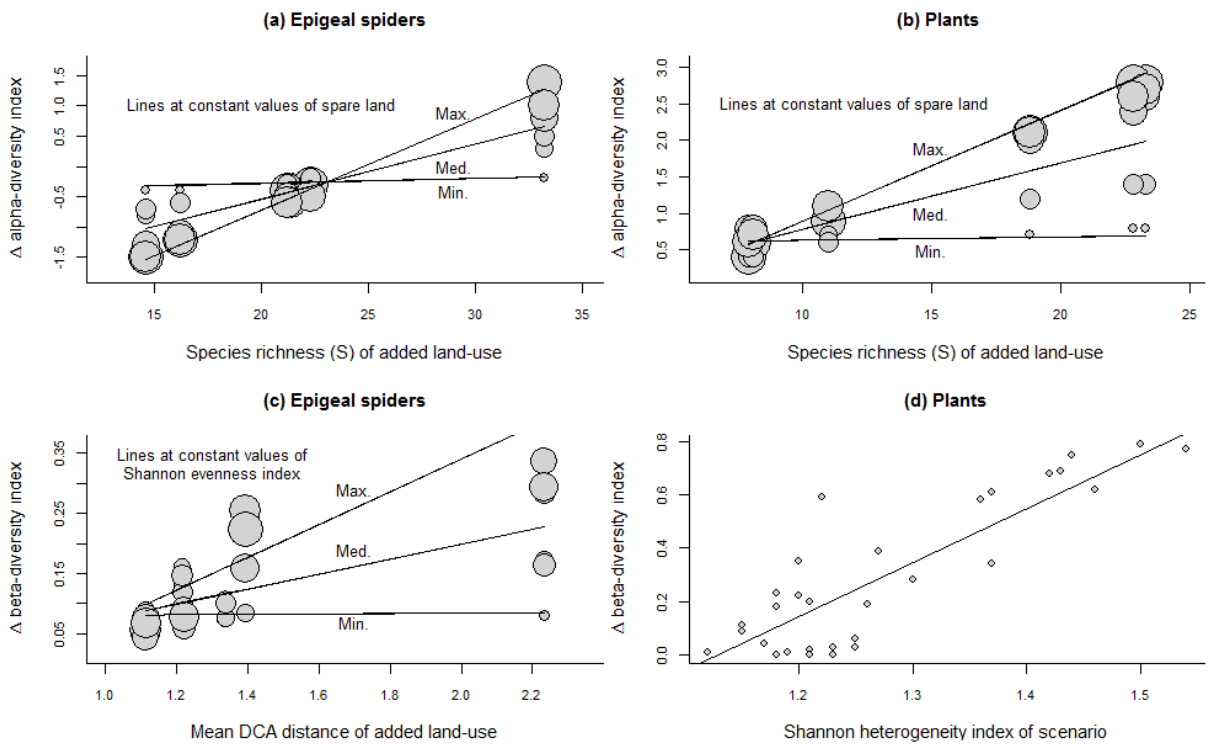
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616   **FIGURE 5. Plots of the best AIC$_c$ models presented in Table 2. For presentation purposes, on (a)-(c)**

617   **circle size is indicative of the relative value on a z-axis (in [a] and [b] this represents the quantity of**

618   **spare land available, and in [c] this represents the natural logarithm of the Shannon land-use**

619   **evenness index of the scenario). Interactions on (a)-(c) have been indicated by selecting three**

620   **constant values on the z-axis (the minimum, median and maximum), and showing the cross-**

621   **section of the modelled plane at that value. Derivation of alpha- and beta-diversity indices from**

622   **field-scale biodiversity data are described in the text.**
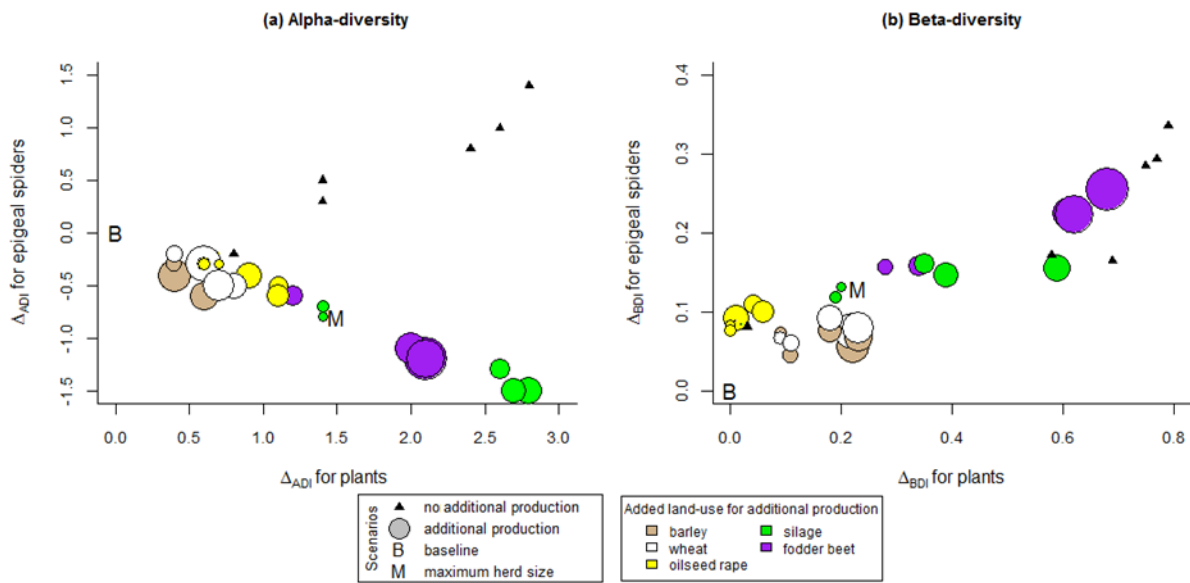
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632 **FIGURE 6. Plots of diversity indices, (a) Δ_{ADI} and (b) Δ_{BDI}, for spiders and plants for spare-land**

633 **scenarios (Fig. 1d). For 'additional production' scenarios, the estimated additional metabolisable**

634 **energy (ME) is indicated by the relative radius of the circle (max for both plots is 1,783 GJ dry**

635 **matter). The 'no additional production' scenarios (addition of extensive grassland to spare land)**

636 **do not have additional ME so are shown by ▲. For comparison, the baseline scenario (Fig. 1a) and**

637 **max. herd size scenario (Fig. 1c) are shown by 'B' and 'M' respectively. Derivation of alpha- and**

638 **beta-diversity indices from field-scale biodiversity data are described in the text.**

Table 1. Fixed effects included in model sets for investigating drivers of alpha (Δ_{ADI}) and beta (Δ_{BDI}) diversity indices of scenarios relative to the baseline scenario (Fig. 1a).

Code	Description	Δ_{ADI}	Δ_{BDI}
SPARE	Area (ha) of spare land available in the test-system	X	X
HETER	Shannon heterogeneity index of the land-use composition of the scenario	X	X
EVEN	Shannon evenness index of the land-use composition of the scenario	X	X
LANDUSES	Number of land-uses within the scenario (5 or 6)	X	X
COVER	Initial area (ha) in the test-system for the land cover replacing the spare land component	X	X
RICHNESS	Mean estimated S of the given taxa of the land-use replacing the spare land component (see Fig. 2)	X	
DCA	Mean Euclidean detrended correspondence analysis distance for the given taxa of the land-use replacing the spare-land component against each other land-use (including itself)		X

	Epigeal spiders					Plants				
DIVERSITY	MODEL	k	AICc	ΔAICc	w_i	MODEL	k	AICc	ΔAICc	w_i
Alpha	**RICH*SPARE**	**4**	**-50.4**	**0.0**	**>0.99**	**RICH*SPARE**	**4**	**-46.3**	**0.0**	**>0.99**
(Δ_{ADI})	RICH*EVEN	4	20.0	70.4	<0.01	RICH*EVEN	4	50.3	96.6	<0.01
	....10 models...					...11 models...				
	[NULL]	1	50.5	100.9	<0.01	[NULL]	1	69.4	115.7	<0.01
	... 4 models...					...3 models...				
Beta	**DCA*EVEN**	**4**	**-117.3**	**0.0**	**0.87**	**HETER**	**2**	**-25.8**	**0.0**	**0.71**
(Δ_{BDI})	**DCA*HETER**	**4**	**-112.4**	**4.8**	**0.08**	**EVEN**	**2**	**-22.33**	**3.5**	**0.13**
	DCA*SPARE	4	-111.6	5.6	0.05	**DCA+HETER**	**3**	**-20.57**	**5.2**	**0.05**
	...4 models...					**DCA+EVEN**	**3**	**-20.53**	**5.3**	**0.05**
	[NULL]	1	-93.22	24.0	<0.01	DCA*EVEN	4	-19.9	5.9	0.04
	... 9 models...					...3 models...				
						[NULL]	1	-3.5	22.3	<0.01
						...8 models...				