

Scotland's Rural College

Routes to achieving sustainable intensification in simulated dairy farms - the importance of production efficiency and complimentary land uses

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1 **Routes to achieving sustainable intensification in simulated dairy farms - the importance of**
2 **production efficiency and complimentary land uses**

3

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20 **ABSTRACT**

- 21 1. Sustainable intensification (SI) is a global challenge, aiming to increase food production
22 whilst conserving biodiversity and ecosystem services. This is contrary to the observed trend
23 of agricultural intensification degrading environmental quality. We developed a framework
24 integrating animal nutrition, crop yields, and biodiversity modelling to explore SI potential in
25 multiple model dairy farming systems through varying crop composition to provide cattle
26 feed rations. We then identified key drivers of biodiversity gain that may be applicable at a
27 wider scale.
- 28 2. We developed multiple feed rations to meet the nutritional demands of a high-yielding,
29 housed dairy herd. The land area required varied due to productivity and nutritional
30 differences between crops, generating spare land. We used published biodiversity models to
31 compare alpha- and beta-diversity of spiders and plants across 36 scenarios that used the
32 spare land in different ways, for either biodiversity maximisation or additional production.
- 33 3. Alpha and beta-diversity for both taxa was greatest in scenarios that maximised spare land
34 and utilised this for species-rich extensive grassland. However, commensurate biodiversity
35 gains for plant alpha-diversity, and spider and plant beta-diversity (respectively 100%, 76%
36 and 86% gain relative to that optimal scenario) were achievable when spare land was used
37 for additional crop production.
- 38 4. Maximising compositional heterogeneity and adding complementary productive land uses to
39 spared land were key to increasing production and beta-diversity, while adding species-rich
40 productive land uses drove increasing production and alpha-diversity.
- 41 5. *Synthesis and applications.* This study indicates the potential for the sustainable
42 intensification (SI) of dairy farming systems through the manipulation of feed rations to
43 increase land efficiency and spare land, which could then be used to enhance production
44 and biodiversity. The optimum land composition depends on target goal(s) (e.g. maximising

45 production and/or biodiversity). Greatest 'win-wins' can be achieved by making land cover
46 more diverse and selecting crops that complement each other in the species they support;
47 highlighting the important role of heterogeneity in the crop matrix. Our study provides a
48 framework that integrates agricultural production efficiency and biodiversity modelling to
49 explore potential routes to achieve SI goals.

50

51 **KEYWORDS**

52 agriculture, biodiversity, dairy, ecological intensification, farming, heterogeneity, land use,
53 sustainable intensification

54

55

56 **INTRODUCTION**

57 The challenge of protecting biodiversity and ecosystem services in the face of increasing food
58 production has led to the call for sustainable intensification (SI) of agricultural land; that is, the
59 simultaneous increase of food production and reduction of environmental pressure (Tilman *et al.*
60 2011; Garnett *et al.* 2013). This is now a key strategic objective of the UN's Food and Agriculture
61 Organisation (FAO 2015). A premise of SI is that increased production should be achieved on existing
62 farmland, because clearance of other habitats for agriculture creates greater environmental
63 degradation and biodiversity loss (Tilman *et al.* 2011; Garnett *et al.* 2013). Thus, a question central to
64 SI is whether we can increase production of existing land while maintaining or improving its
65 biodiversity.

66

67 **Livestock production and biodiversity**

68 Demand for meat and dairy products have increased substantially, with global cattle stock increasing
69 by 57% between 1961 and 2016 (FAOSTAT 2017). Grassland management has been intensified,
70 changing the functional composition of vegetation, with associated global biodiversity declines being
71 observed across taxa (Plantureux, Peeters & McCracken 2005). Functional and/or taxonomic
72 changes to vegetation composition have benefitted some ecosystem services (e.g. food provisioning
73 / nutrient cycling services), at the cost of others (e.g. regulating, cultural and biodiversity services)
74 (Allan *et al.* 2015).

75

76 To satisfy the nutritional demands (i.e. fat, carbohydrate, protein) of high-yielding dairy cows (≥ 40 L
77 milk d^{-1} animal $^{-1}$), feed rations comprise of multiple crops (Toma *et al.* 2013). Variation in the
78 composition of animal feeds and in-crop productivity produces large differences in the amount and
79 composition of land required to fulfil the dietary needs of a given herd at a target milk yield. In the
80 context of SI, this creates a dual opportunity: (1) if land-use composition can be modified to increase

81 production, modifications may be steered towards productive land compositions that maintain, or
82 even improve, biodiversity; (2) if efficiency improvements mean land-use composition can be
83 modified to produce spare land, whilst still maintaining or increasing production, this land might be
84 used for high biodiversity land-uses. Grassland simulations indicate heterogeneity of management
85 intensity can be varied to simultaneously increase arthropod populations and food production
86 (Simons & Weisser 2017). However, this has not been explored in mixed land-use systems where
87 there may be greater scope to manipulate crop composition to achieve multiple benefits.

88

89 Our study integrates animal nutrition, crop productivity, and biodiversity modelling to explore a
90 range of land-use scenarios that meet the feed requirements of a model dairy system to determine
91 the possibility of increasing food production whilst maintaining or enhancing biodiversity. We
92 developed a range of scenarios and manipulated the composition of land-covers to determine where
93 'win wins' could be achieved and to identify key drivers of production and/or biodiversity gains. All
94 scenarios provided feed for a herd of at least 100 permanently housed dairy cattle but differed in
95 terms of their land-use composition and thus land-efficiency (Fig. 1). More land-efficient scenarios
96 generated up to 18% 'spare land', which could then be used for additional production, or allocated
97 to a biodiversity-rich habitat (i.e. extensive grassland). Extending published land-use/biodiversity
98 models, we estimated indices for scenarios' alpha and beta-diversity for two functional groups
99 (plants and epigeal spiders) where extensive biodiversity data were available (Downie *et al.* 1999;
100 Wilson *et al.* 2003). We hypothesised that production of the system could be maintained or even
101 improved, whilst also improving or maintaining diversity of plants and spiders. We thus explore the
102 potential for SI within a dairy system and discuss implications for achieving SI goals more widely.

103

104 **MATERIALS AND METHODS**

105 **Land-use scenarios**

106 Our modelling process explored a 'home-grown feeds' dairy system, where nutritional requirements
107 of permanently-housed dairy cows are grown on the farm (Roberts & March 2014). We used an
108 industry-standard livestock nutrition feeding model, FeedByte (Scottish Agricultural College 2006), to
109 design alternative feed rations for a model herd of 100 Holstein-Friesian cows with a target milk
110 yield of 9,500 L yr⁻¹ (equivalent to 40 L of milk cow⁻¹ day⁻¹ including an unproductive period). This
111 commercially-used software uses least-cost diet formulation and linear-programming and is used by
112 industry and researchers to design real feed rations sufficient for livestock herds (Chagunda,
113 Flockhart & Roberts 2010). Rations needed to comprise of grass silage, barley, wheat, oilseed rape
114 and fodder beet as comprehensive biodiversity data existed for these crops and they varied with
115 respect to the availability of specific nutrients. Each crop alone would not provide the combination
116 of nutrients (e.g. proteins, carbohydrates or fats) required for high-yielding cattle. For each ration,
117 we used productivity estimates using both fresh-weight and dry-matter (DM) yields (kg ha⁻¹) as well
118 as nutritive value estimates (g kg⁻¹) for the constituting crops (Supporting Information Table S1) to
119 calculate the land area required. Due to differences in crop yields and nutritive values, the amount
120 of land required to meet the herds' nutritional requirements differed substantially between rations.
121 The feed ration requiring the most land for our target milk yield and herd size (i.e. the least land-
122 efficient ration) was designated the 'baseline scenario' (103.8 ha) (Fig. 1a). Our most land-efficient
123 feed ration provided the same milk yield and herd size on less land (88.0 ha). Consequently through
124 manipulating the quantities of different crops in the feed ration we could generate up to 15.8 ha
125 (18%) of spare land relative to the baseline. This generated our maximum spare land test-system
126 (Fig. 1b).

127

128 To compare a variety of scenarios where feed rations are met via different land areas and
129 compositions, we generated five intermediate test-systems varying in land-efficiency between the
130 baseline scenario and the max. spare land test-system (1%, 6%, 8%, 12% and 15% spare land c.f. the

131 baseline). The result was six test-systems (Fig. 1b) containing spare land; the choice of six allowed
132 generation of 36 'spare-land scenarios', considered a suitable sample size for exploring drivers of
133 biodiversity change across scenarios. We generated spare-land scenarios by replacing the spare land
134 component of each test-system (Fig. 1b) with one of six land uses (grass silage, barley, wheat,
135 oilseed rape, fodder beet or extensive grassland) (Fig. 1d). Spare land scenarios thus differed in both
136 the amount of spare land and the land use replacing the spare land component. Spare land scenarios
137 could be divided into:

138

139 (1) 'additional production' scenarios where spare land was used to grow more of one of the
140 productive crops (silage, wheat, barley, oilseed rape or fodder beet; Fig. 1d i).

141

142 (2) 'no additional production' scenarios where spare land was designated to extensive grassland (Fig.
143 1d ii). Of the land uses available in our empirical datasets, we selected extensive grassland to
144 represent a high-biodiversity, low/non-productive land use with no improvement or cutting regime.

145

146 Sparing land from production and designating to biodiversity-rich habitats, as in our 'no additional
147 production' scenarios, is a frequent focus of agri-environment prescriptions (e.g. fallow, floristically
148 diverse field margins). However, the original datasets we used (see below) did not contain data from
149 such habitats so we used data from extensively grazed, semi-natural grassland (e.g. calcifugous and
150 *Juncus*-dominated) to represent a low-production, high biodiversity land use as our alternative
151 strategy to increasing production. These are not directly equivalent to newly created agri-
152 environment habitats, since environmental (e.g. topography, altitude, soil, climate) and socio-
153 ecological (grazing regimes and underlying productivity of the land) constraints would have
154 historically prevented intensification on such areas with long-term extensive management resulting

155 in unique, species-rich, communities (Downie *et al.* 1999; Wilson *et al.* 2003; Cole *et al.* 2005). We
156 discuss the implications of this for our findings below.

157

158 We also explored an additional scenario where the desired outcome was to maximise milk
159 production ('max. herd size': Fig. 1c). This was achieved by scaling up the land composition of the
160 most land-efficient test-system, which had 18% spare land ('max. spare land'; Fig. 1b), to occupy the
161 area of the baseline scenario resulting in an increase from 100 to 118 cattle and thus increasing milk
162 production by 18%. Thus we generated 38 scenarios in total: the baseline scenario (Fig. 1a), the
163 max. herd size scenario (Fig. 1c), and the 36 spare land scenarios (Fig. 1d).

164

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

166 For each of the 36 spare land scenarios (Fig. 1d) and the max. herd size scenario (Fig. 1c), we
167 calculated indices of alpha and beta-diversity for plants and spiders, and compared these with the
168 baseline scenario. This analysis framework is illustrated in Fig. 2. Diversity indices were generated
169 from published biodiversity models derived from field studies measuring plant and spider species
170 richness in the agricultural land-covers in our system (Downie *et al.* 1999; Wilson *et al.* 2003).

171

172 Data had been collected in nine geographical locations in Scotland ranging from intensive arable and
173 grassland landscapes, mixed farming and crofting. Vegetation surveys were conducted over three
174 years (1995-1997) at 87 sites with data being collected from permanent 10×10 m quadrats on three
175 occasions during peak growing season (June-September) (Wilson *et al.* 2003). Epigeal spiders were
176 surveyed over two years (1996-1997) at a total of 71 sites using a 16 m transect of nine pitfall traps
177 (75 mm diameter and 100 mm deep). Pitfall trapping was conducted May-September with contents
178 collected monthly (Downie *et al.* 1999).

179

180 Plants and spiders are key components of agricultural ecosystems, strongly driven by land
181 management (Wilson *et al.* 2003; Batáry *et al.* 2012). Plants are key drivers of invertebrate
182 biodiversity (Symstad, Siemann & Haarstad 2000) and provide shelter and breeding sites for many
183 species, while spiders are important polyphagous predators contributing to natural pest control
184 (Herzog *et al.* 2013). Alongside wild bees and earthworms, they provide appropriate and robust
185 metrics for measuring agricultural biodiversity (Herzog *et al.* 2013).

186

187 To generate alpha-diversity estimates for spiders and plants we used species richness (S) models
188 fitted by Downie *et al.* (1999) and Wilson *et al.* (2003). These models were developed to explore
189 drivers of field-scale S in a range of land uses. The process we used to generate an alpha-diversity
190 index (ADI) for each scenario is described in detail in Appendix S2. Briefly, the ADI for each scenario
191 is defined as the mean predicted field-scale S across land-uses, weighted by the proportional area of
192 each land-use. Weighted mean S is a recognised metric to explore impacts of agricultural land-use
193 composition on biodiversity (e.g. Hiron *et al.* 2015).

194

195 To generate beta-diversity indices we used data describing the assemblage structure of our target
196 groups collected by Downie *et al.* (1999) and Wilson *et al.* (2003). Beta-diversity here describes the
197 extent of variation of species assemblages between different land-uses. Raw species data for the
198 models considered in the ADI analyses were not available. However, detrended correspondence
199 analysis (DCA) axes scores derived from the raw data were available (Murphy *et al.* 1998). The
200 distance between two sites in a DCA ordination provides a measure of similarity in species
201 composition, with smaller distances indicating greater similarity (Smol *et al.* 2005). The process we
202 used to generate beta-diversity indices (BDI) for each scenario is described in detail in Appendix S2.
203 Briefly, the BDI for each scenario is defined as the median DCA distance between random pairs of

204 land-uses within each scenario, weighted by the proportional area of each land-use. We provide R
205 code for the estimation of ADI and BDI in Appendix S3

206

207 While each spare-land scenario (Fig. 1d) maintained our target milk yield, a scenario's 'value' can be
208 modified relative to the baseline scenario in two main ways: (i) biodiversity value, indicated by ADI
209 and BDI; and (ii) production value, which can be increased when the spare-land component is used
210 for additional production. To determine how spider and plant ADI and BDI changed relative to the
211 baseline scenario we subtracted the baseline ADI and BDI values from those calculated for each
212 scenario, the results termed Δ_{ADI} and Δ_{BDI} (Fig. 2).

213

214 To allow us to simultaneously explore both the change in biodiversity and production for additional
215 production scenarios, we estimated the production capacity of the spare land component as
216 metabolisable energy ('ME', MJ kg⁻¹ DM) to standardise this benefit across different crops. ME
217 provided a more direct measure of production capacity than monetary value of crops, which is
218 dependent on a range of additional market factors. We derived ME of the spare land component for
219 each additional production scenario by multiplying industry estimates of ME by yield (kg DM ha⁻¹)
220 (SAC Consulting 2014) and area (ha). The baseline scenario did not produce any additional energy on
221 top of feeding the herd of 100 cattle, and consequently the calculated ME value reflects change
222 relative to the baseline (i.e. Δ_{ME}).

223

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

225 To investigate which factors drove the magnitude of Δ_{ADI} or Δ_{BDI} for each taxa, we generated four
226 candidate sets of general linear mixed models, with Δ_{ADI} or Δ_{BDI} for plants and spiders as the
227 response variables. Each spare land scenario provided a single estimate of Δ_{ADI} and Δ_{BDI} (Fig. 2) so the
228 sample size for the analysis was 36. We included test-system identity (Fig. 1b), and land-use

229 constituting the spare-land component (Fig. 1d) as random effects. We describe the fixed effects
230 included in our models in Table 1.

231

232 For each taxa (plant/spider) and response variable ($\Delta_{ADI}/\Delta_{BDI}$) we compared models using a small-
233 sample Akaike's Information Criterion (AICc) and Akaike's weights (w_i) (Johnson & Omland 2004).

234 We included several multivariate models and the appropriate null model. We restricted inference on
235 the key drivers of Δ_{ADI} or Δ_{BDI} to examining the smallest set of models containing a summed $w_i \geq 0.90$.

236

237 **RESULTS**

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

239 Predicted estimates of S for spiders and plants from the alpha-diversity models for individual land-
240 uses are shown in Figs. 3a-b. Predictions of spider S were as low as 11.7 (lower 95% confidence limit
241 [CL₉₅] for silage) and as high as 41.9 (upper CL₉₅ for extensive grassland). For plants, values ranged
242 from 6.1 species (lower CL₉₅ for barley) to 29.6 (upper CL₉₅ for extensive grassland). These were
243 within the observed field-scale ranges from the original data-sets of 10-56 spider species (Downie *et*
244 *al.* 1999) and 5-57 plant species (Wilson *et al.* 2003).

245

246 Assemblage structure of spider and plant communities was most similar in oilseed rape and cereals
247 (i.e. barley and wheat) with communities in these crops showing the greatest disparity with those in
248 extensive grasslands and fodder beet (Figs. 3c-d).

249

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

251 The baseline scenario had a predicted ADI (the mean field-scale S across land-uses, weighted by the
252 proportional area of each land-use) of 19.8 (CL₉₅ 17.8-22.8) spider species and 13.4 (CL₉₅ 11.9-15.2)
253 plant species. Δ_{ADI} and Δ_{BDI} for each taxa and scenario are shown in Fig. 4. In all cases, 'no additional

254 production' scenarios with maximum spare land gave the highest biodiversity benefit (labelled 'NAP'
255 in Fig. 4). However, for both taxa Δ_{BDI} , and for plant Δ_{ADI} , additional production scenarios existed that
256 generated similar biodiversity gains to the latter (optimal scenarios labelled 'AP' in Fig. 4). For
257 spiders, additional production scenarios did not enhance ADI due to the baseline scenario having a
258 relatively high ADI (Fig. 4a).

259

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

261 For both spider and plant Δ_{ADI} , a single model was in the confidence set (Table 2). This contained the
262 estimated S of the added land-use, the area of spare land, and their interaction (Figs. 5a-b). Adding a
263 land-use with a comparatively high estimated S to the spare-land component increased Δ_{ADI} , and this
264 interacted positively with amount of spare land.

265

266 For spider Δ_{BDI} , two models were in the confidence set (Table 2), containing the mean DCA distance
267 for the land-use being added, the land-use evenness ($w_i = 0.87$) or heterogeneity ($w_i = 0.08$), and
268 their interaction. A higher mean DCA distance of the land-use being added increased Δ_{BDI} , but only
269 where land-use evenness was also increased (Fig. 5c). For plant Δ_{BDI} , four models were in the
270 confidence set (Table 2). The two best models contained either the land-use heterogeneity ($w_i =$
271 0.71) or evenness ($w_i = 0.13$), showing positive relationships (Fig. 5d).

272

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

274 For both Δ_{ADI} (Fig. 6a) and Δ_{BDI} (Fig. 6b) utilising spare land as extensive grassland delivered the
275 optimal biodiversity gain across taxa, but generated no Δ_{ME} gain. For Δ_{BDI} , not only was a max. spare
276 land + fodder beet scenario able to achieve gains almost commensurate with the best no additional
277 production scenario for both spiders and plants, but it did so while achieving the highest Δ_{ME} (Fig.
278 6b).

279

280 For Δ_{ADI} , the relatively high ADI value for spiders in the baseline scenario resulted in a complex
281 picture (Fig. 6a). All additional production scenarios showed a reduced ADI for spiders, but increased
282 ADI for plants. Some added fodder beet scenarios achieved relatively high plant Δ_{ADI} and moderate
283 Δ_{ME} gains but at a cost to spider Δ_{ADI} values (albeit with spider losses lower in magnitude than plant
284 gains). Some added silage scenarios achieved greater plant Δ_{ADI} gains than added fodder beet (equal
285 to plant Δ_{ADI} gain in the best no additional production scenario) but at a cost to spider Δ_{ADI} and a
286 lower Δ_{ME} , indicating trade-offs between potential gains.

287

288 **DISCUSSION**

289 **Sustainable intensification outcomes**

290 Simultaneous gains in production and biodiversity were simulated in our model dairy system
291 illustrating the potential for SI. We achieved the biggest simultaneous gains in beta-diversity and
292 production by increasing the system's efficiency to feed the dairy herd using a smaller area of land,
293 and allocating the spare land generated to a productive crop that complemented those already
294 present in the system with respect to the species supported. For plant alpha-diversity, we achieved
295 highest gains in production and species richness by allocating the spare land to a species-rich,
296 productive land use.

297

298 Our simulations challenge the well-observed negative relationship between productivity and
299 biodiversity in agriculture (Krebs *et al.* 1999). SI has been demonstrated in small farms in developing
300 countries using a combination of approaches including integrated plant nutrient systems, no-
301 till/conservation agriculture and integrated pest management (Chappell & LaValle 2011). Evidence of
302 SI in highly-productive agricultural systems is less well documented with gains in biodiversity
303 typically being accompanied by yield losses (Gabriel *et al.* 2010; Firbank *et al.* 2013). SI appears

304 particularly challenging in intensive dairy farms - a study exploring innovative management practices
305 found dairy farms notable in their failure to achieve simultaneous production and biodiversity gains
306 (Firbank *et al.* 2013). Our results highlight a potential route to SI in home-grown dairy systems.

307

308 With increasing pressure on our finite agricultural land, SI is important both for local and wider scale
309 biodiversity outcomes. Our 'no additional production' scenarios, increased farm-scale biodiversity,
310 but without increasing production. Increasing demand for food would therefore have to be met by
311 intensification of other farmland and/or generating new farmland (see Fischer *et al.* 2014), with
312 potentially negative biodiversity implications at a wider scale. Our 'additional production' scenarios
313 that achieved SI would not only benefit local biodiversity but also reduce production pressure on
314 land elsewhere. Ultimately, SI solutions will be important in allowing biodiversity to be increased
315 both on a farm scale through land-sharing, and by increasing production on existing farmland which
316 could protect biodiversity-rich areas at the landscape scale ('land-sparing'). The land sharing/sparing
317 debate, however, can overlook the wide range of ecosystem services agricultural land can provide
318 (Fischer *et al.* 2014) and measures that promote biodiversity often improve landscape multi-
319 functionality enhancing a wide-range of ecosystem services (Allan *et al.* 2015).

320

321 **Potential for implementation to real systems**

322 A key finding was the important role that crop heterogeneity played in increasing beta-diversity and
323 production. Loss of heterogeneity (of both semi-natural and agricultural habitats) is a key driver of
324 biodiversity declines, and its restoration represents a mitigation strategy (Benton, Vickery & Wilson
325 2003). Agricultural policies aimed at stemming biodiversity loss typically incentivise farmers to
326 increase heterogeneity via agri-environment schemes (AES), which may involve in-production (land-
327 sharing) or out-of-production (land-sparing) approaches (Batáry *et al.* 2015). Restoring semi-natural
328 habitats in intensive agricultural landscapes typically involves removing land from production, to

329 increase spatial and structural heterogeneity (Ovenden, Swash & Smallshire 1998). Our results
330 suggest that incentivising farmers to increase heterogeneity of their productive land could lead to
331 biodiversity gains whilst increasing production. At a regional level, that could involve a coordinated
332 approach to produce crops that have high complementarity and are not already dominant.

333

334 In introducing a crop diversification component within its compulsory greening measures (EU
335 Regulation 1307/2013), the EU's 2014 Common Agricultural Policy reform may represent a step in
336 the right direction. However, 'diversification' in this legislation narrowly focuses on compositional
337 heterogeneity, bringing its effectiveness under question. Josefsson et al. (2017) found that crop
338 composition *per se* did not influence bird species richness, but structural crop heterogeneity did
339 have a positive effect. For such measures to make a real impact, they may need to go further and be
340 backed-up by a political will to diversify farmland. Some AES are piloting a cluster farming approach
341 to provide landscape-scale benefits (e.g. Natural England 2017); SI strategies could also potentially
342 operate at a multi-farm scale.

343

344 With SI goals aiming to increase production and thus income, appropriate knowledge exchange may
345 enhance uptake of SI-focussed management without additional incentives, although in the UK some
346 innovative farms seen to achieve SI outcomes had relied on AES subsidies to enhance biodiversity
347 (Firbank *et al.* 2013). Improving links between researchers, advisors and farmers are identified as
348 important for uptake of SI practices in developing countries (Pretty, Toulmin & Williams 2011).
349 Promoting biodiversity can have direct economic benefits through enhancing ecosystem services
350 such as natural pest control and pollination, potentially contributing to SI via ecological
351 intensification (Tscharrntke *et al.* 2012). Promoting the potential economic benefits of enhancing
352 biodiversity may also be important (Pywell *et al.* 2015). For example, insect pollination can increase

353 yield of oilseed rape (Bommarco, Marini & Vaissière 2012) while the presence of natural enemies of
354 aphids can increase yield of barley (Östman, Ekbom & Bengtsson 2003).

355

356 Incentivising heterogeneity requires careful consideration of costs and benefits. Crop heterogeneity
357 may promote economic sustainability, providing insurance against unpredictable growing conditions,
358 pest or disease outbreaks, or market variability (Garnett *et al.* 2013). In our home-grown system, the
359 production of additional crops not directly required for *in situ* cattle, requires additional market
360 engagement, but markets can be unpredictable. Growing new crops may also incur direct costs, such
361 as those for new machinery or alternative agrochemicals. Our simple measure of increased
362 production was solely based on the energetic yield of a crop, while demand depends on other socio-
363 economic drivers such as available income and consumer behaviour (Valin *et al.* 2014).

364

365 Simultaneously enhancing biodiversity and production clearly requires incorporating complex
366 information derived from disparate sources. For example, here we integrated biodiversity data from
367 both published literature, dietary modelling that combines nutritional equations to determine
368 protein and energy requirements, and finally crop production data. Farmers are familiar with a range
369 of decision-support tools, such as nutrition models, yield estimates, agronomist advice and
370 guidelines on incentives for AES. Comprehensive biodiversity datasets that evaluate biodiversity
371 across habitats are, however, often only available as summary information in scientific publications
372 (e.g. Cole *et al.* 2017). This could make it difficult for farmers to adequately consider biodiversity in
373 decision-making processes. There is great potential for existing monitoring schemes (e.g. UK's
374 Breeding Bird Survey: <https://www.bto.org/volunteer-surveys/bbs>; UK's Butterfly Monitoring
375 Scheme: <http://www.ukbms.org/>) to assist in the collection of comprehensive biodiversity data
376 across taxa and land covers, but more fundamental is providing the resultant data to land managers
377 in a usable format. Our framework highlights the potential for current databases (e.g. crop

378 productivity and land-cover/biodiversity) to be integrated to create a SI decision-support tool. In
379 addition to including production and biodiversity outcomes, such a tool could also include other
380 environmental (e.g. reducing greenhouse gas emissions, mitigating diffuse pollution) and agronomic
381 (e.g. reduced agrochemical applications) benefits.

382

383 Simulation studies are important first steps in assessing potential SI outcomes (e.g. Simons &
384 Weisser 2017). Ultimately, however, model predictions require trialling in real landscapes. While
385 simulated studies provide data-driven working hypotheses, they have inevitable limitations. For
386 example, we considered compositional but not configurational heterogeneity, which can drive
387 biodiversity patterns in birds (Hiron *et al.* 2015). In addition, data constraints meant that to
388 represent high-biodiversity, low-input habitats in our 'no additional production' scenarios we used
389 data from historical extensively grazed grasslands rather than potentially more appropriate AES
390 prescriptions (i.e. habitats formed from previously cultivated land that have not had a history of low-
391 input management). Extensive grasslands support unique species assemblages, and it is unlikely that
392 AES prescriptions (e.g. species-rich field margins) would reach the potential biodiversity value of
393 such grasslands (Downie *et al.* 1999; Wilson *et al.* 2003). Our 'no additional production scenarios'
394 may therefore over-estimate biodiversity gains. Agri-environment interventions can, however, not
395 only increase ecological connectivity and provide ecological contrast that enhances resource
396 diversity, stability and availability supporting a wider suite of species (Batáry *et al.* 2015; Cole *et al.*
397 2017), but can also enhance biodiversity-dependant ecosystem services thus benefitting production
398 (Pywell *et al.* 2015). Landscape trials of scenarios could elucidate effects of configurational,
399 structural and compositional heterogeneity, and test biodiversity benefits of AES prescriptions
400 against our assumptions using historical extensively grazed habitats.

401

402 A key reason for investigating potential for SI outcomes in a home-grown dairy system was the
403 closed nature of the system, whereby the land required to feed the herd is *in situ*. Feeding housed
404 cattle is a common dairy production system (e.g. 55% of UK dairy farms house cattle year-round, a
405 percentage which is growing - March *et al.* 2014) but housed cattle can be fed on home-grown or
406 bought-in feeds, or a combination. Home-grown systems represent one end of a spectrum of
407 possible feeding systems for housed dairy cattle (Roberts & March 2014), one which is fully self-
408 sufficient in terms of feeds. While data on the extent of feed self-sufficiency of farms are not widely
409 available, it can be regionally high (e.g. averaging 79-85% in W France: Brocard *et al.* 2016). Higher
410 self-sufficiency of feeds is seen to reduce consumption of non-renewable energy (i.e. reduced
411 transport of bought-in feeds) and raise nutrient efficiency (Gaudino *et al.* 2018). In Europe,
412 producing local sources of protein for dairy cattle is likely to reduce reliance on imported soybean
413 *Glycine max* imports, associated with high greenhouse gas emissions (Hörtenhuber, Lindenthal &
414 Zollitsch 2011), while programs have been established aiming to increase levels of dairy feed self-
415 sufficiency (Ineichen *et al.* 2014). Our simulations demonstrate a further potential environmental
416 benefit of home-grown systems - that composition of feed crops could be managed locally to
417 provide simultaneous productivity and biodiversity gains.

418

419 **Conclusions**

420 Achieving SI is an important but ambitious aim (Tilman *et al.* 2011; Garnett *et al.* 2013). Our study
421 combined biodiversity data, yield data and nutrition models to find routes to simultaneous increases
422 in production and biodiversity in a home-grown dairy system. This was optimised by maximising
423 land-efficiency and targeting additional production to a land cover that had relatively high species
424 richness (alpha-diversity), and complemented species in existing crops (beta-diversity). This
425 highlights the importance of integrating agronomic efficiency, land cover heterogeneity and species
426 richness/complementarity of both productive and non-productive land covers within an SI

427 framework. Where comprehensive biodiversity/production data exist, our framework could be
428 adaptable to other taxonomic groups, production systems and regions. Agri-environment policy
429 should focus not only on increasing the quality and heterogeneity of semi-natural habitats, but also
430 on enhancing agricultural efficiency and the complementarity and heterogeneity of productive land
431 covers. Through developing a framework that integrates crop productivity and biodiversity
432 modelling to seek optimal production-biodiversity scenarios, this study presents a route to identify
433 key drivers of production and biodiversity gain, a key goal of SI, that may be applicable at a wider
434 scale.

435

436 **AUTHORS' CONTRIBUTIONS**

437 All authors developed the initial concepts and MAL designed test scenarios. PJCW conducted
438 analyses and LJC sourced the datasets. PJCW and LJC wrote the initial draft and all authors
439 contributed to revisions and gave final approval for publication.

440

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445

446 **DATA ACCESSIBILITY**

447 Data available from Edinburgh Napier University Repository. DOI: 10.17869/enu.2019.1497843
448 (White *et al.* 2019).

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451 **REFERENCES**

- 452 Allan, E., Manning, P., Alt, F., Binkenstein, J., Blaser, S., Bluthgen, N., Bohm, S., Grassein, F., Holzel,
453 N., Klaus, V.H., Kleinebecker, T., Morris, E.K., Oelmann, Y., Prati, D., Renner, S.C., Rillig, M.C.,
454 Schaefer, M., Schloter, M., Schmitt, B., Schoning, I., Schrumpp, M., Solly, E., Sorkau, E., Steckel,
455 J., Steffen-Dewenter, I., Stempfhuber, B., Tschapka, M., Weiner, C.N., Weisser, W.W., Werner,
456 M., Westphal, C., Wilcke, W. & Fischer, M. (2015) Land use intensification alters ecosystem
457 multifunctionality via loss of biodiversity and changes to functional composition. *Ecology*
458 *Letters*, **18**, 834–843.
- 459 Batáry, P., Dicks, L. V., Kleijn, D. & Sutherland, W.J. (2015) The role of agri-environment schemes in
460 conservation and environmental management. *Conservation Biology*, **29**, 1006–1016.
- 461 Batáry, P., Holzschuh, A., Orci, K.M., Samu, F. & Tschardtke, T. (2012) Responses of plant, insect and
462 spider biodiversity to local and landscape scale management intensity in cereal crops and
463 grasslands. *Agriculture, Ecosystems & Environment*, **146**, 130–136.
- 464 Benton, T.G., Vickery, J.A. & Wilson, J.D. (2003) Farmland biodiversity: is habitat heterogeneity the
465 key? *Trends in Ecology & Evolution*, **18**, 182–188.
- 466 Bommarco, R., Marini, L. & Vaissière, B.E. (2012) Insect pollination enhances seed yield, quality, and
467 market value in oilseed rape. *Oecologia*, **169**, 1025–1032.
- 468 Brocard, V., Jost, J., Rouillé, B., Caillaud, D., Caillat, H. & Bossis, N. (2016) Feeding self-sufficiency
469 levels in dairy cow and goat farms in Western France: current situation and ways of
470 improvement. *Grassland Science in Europe*, **21**, 53–55.
- 471 Chagunda, M.G.G., Flockhart, J.F. & Roberts, D.J. (2010) The effect of forage quality on predicted
472 enteric methane production from dairy cows. *International Journal of Agricultural*
473 *Sustainability*, **8**, 250–256.
- 474 Chappell, M.J. & LaValle, L.A. (2011) Food security and biodiversity: can we have both? An
475 agroecological analysis. *Agriculture and Human Values*, **28**, 3–26.

476 Cole, L.J., Brocklehurst, S., Robertson, D., Harrison, W. & McCracken, D.I. (2017) Exploring the
477 interactions between resource availability and the utilisation of semi-natural habitats by insect
478 pollinators in an intensive agricultural landscape. *Agriculture, Ecosystems & Environment*, **246**,
479 157–167.

480 Cole, L.J., McCracken, D.I., Downie, I.S., Dennis, P., Foster, G.N., Waterhouse, T., Murphy, K.J., Griffin,
481 A.L. & Kennedy, M.P. (2005) Comparing the effects of farming practices on ground beetle
482 (Coleoptera: Carabidae) and spider (Araneae) assemblages of Scottish farmland. *Biodiversity
483 and Conservation*, **14**, 441–460.

484 Downie, I.S., Wilson, W.L., Abernethy, V.J., McCracken, D.I., Foster, G.N., Ribera, I., Murphy, K.J. &
485 Waterhouse, A. (1999) The impact of different agricultural land-uses on epigeal spider diversity
486 in Scotland. *Journal of Insect Conservation*, **3**, 273–286.

487 FAO. (2015) *FAO and the 17 Sustainable Development Goals*. Food and Agriculture Organisation of
488 the United Nations, Rome.

489 FAOSTAT. (2017) FAOSTAT: Food & Agriculture Organization of the United Nations Statistic Division,
490 <http://faostat3.fao.org/browse/Q/QA/E>

491 Firbank, L.G., Elliott, J., Drake, B., Cao, Y. & Gooday, R. (2013) Evidence of sustainable intensification
492 among British farms. *Agriculture, Ecosystems & Environment*, **173**, 58–65.

493 Fischer, J., Abson, D.J., Butsic, V., Chappell, M.J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H.G.
494 & von Wehrden, H. (2014) Land Sparing Versus Land Sharing: Moving Forward. *Conservation
495 Letters*, **7**, 149–157.

496 Gabriel, D., Sait, S.M., Hodgson, J.A., Schmutz, U., Kunin, W.E. & Benton, T.G. (2010) Scale matters:
497 the impact of organic farming on biodiversity at different spatial scales. *Ecology Letters*, **13**,
498 858–869.

499 Garnett, T., Appleby, M.C., Balmford, A., Bateman, I.J., Benton, T.G., Bloomer, P., Burlingame, B.,
500 Dawkins, M., Dolan, L., Fraser, D., Herrero, M., Hoffmann, I., Smith, P., Thornton, P.K., Toulmin,

501 C., Vermeulen, S.J. & Godfray, H.C.J. (2013) Sustainable intensification in agriculture: premises
502 and policies. *Science*, **341**, 33–4.

503 Gaudino, S., Reidsma, P., Kanellopoulos, A., Sacco, D., van Ittersum, M., Gaudino, S., Reidsma, P.,
504 Kanellopoulos, A., Sacco, D. & van Ittersum, M.K. (2018) Integrated Assessment of the EU's
505 Greening Reform and Feed Self-Sufficiency Scenarios on Dairy Farms in Piemonte, Italy.
506 *Agriculture*, **8**, 137.

507 Herzog, F., Jeanneret, P., Ammari, Y., Angelova, S., Arndorfer, M., Bailey, D., Balazs, K., Báldi, A.,
508 Bunce, R.G.H., Choisis, J.-P., Dennis, P., Dyman, T., Eiter, S., Fjellstad, W., Fraser, M.D., Frank, T.,
509 Friedel, J.K., Garchi, S., Geijzendorffer, I., Gomiero, T., Jerkovich, G., Jongman, R.G.H., Kainz, M.,
510 Kakudidi, E., Kelemen, E., Kolliker, R., Kwikiriza, N., Kovács-Hostyánszki, A., Last, L., Luscher, G.,
511 Moreno, G., Nkwiine, C., Opio, J., Oschatz, M.-L., Paoletti, M.G., Pointereau, P., Sarthou, J.-P.,
512 Schneider, M.K., Siebrecht, N., Sommaggio, D., Stoyanova, S., Targetti, S., Viaggi, D., Wolfrum,
513 S. & Yashchenko, S. (2013) Measuring farmland biodiversity. *Solutions*, **4**, 52–58.

514 Hiron, M., Berg, Å., Eggers, S., Berggren, Å., Josefsson, J. & Pärt, T. (2015) The relationship of bird
515 diversity to crop and non-crop heterogeneity in agricultural landscapes. *Landscape Ecology*, **30**,
516 2001–2013.

517 Hörtenhuber, S.J., Lindenthal, T. & Zollitsch, W. (2011) Reduction of greenhouse gas emissions from
518 feed supply chains by utilizing regionally produced protein sources: the case of Austrian dairy
519 production. *Journal of the Science of Food and Agriculture*, **91**, 1118–1127.

520 Ineichen, S., Piccand, V., Chevalley, S., Reidy, B. & Cutullic, E. (2014) Feeding strategies and feed self-
521 sufficiency of dairy farms in the lowland and mountain area of Western Switzerland. *Grassland
522 Science in Europe*, **19**, 680–682.

523 Johnson, J.B. & Omland, K.S. (2004) Model selection in ecology and evolution. *Trends in Ecology and
524 Evolution*, **19**, 101–108.

525 Krebs, J.R., Wilson, J.D., Bradbury, R.B. & Siriwardena, G.M. (1999) The second Silent Spring? *Nature*,

526 **400**, 611–612.

527 March, M.D., Haskell, M.J., Chagunda, M.G.G., Langford, F.M. & Roberts, D.J. (2014) Current trends
528 in British dairy management regimens. *Journal of Dairy Science*, **97**, 7985–7994.

529 Murphy, K.J., McCracken, D.I., Foster, G.N., Waterhouse, A., Furness, R., Abernethy, V.J., Downie,
530 I.S., Wilson, W.L., Adam, A. & Ribera, I. (1998) *Functional Analysis of Plant-Invertebrate-Bird*
531 *Biodiversity on Scottish Agricultural Land. Final Report to SOAEFD No: UGW/814/94*, 51.
532 University of Glasgow, Glasgow.

533 Natural England. (2017) Farmer Clusters - working together to achieve more,
534 [https://naturalengland.blog.gov.uk/2017/03/21/farmer-clusters-working-together-to-achieve-](https://naturalengland.blog.gov.uk/2017/03/21/farmer-clusters-working-together-to-achieve-more/)
535 more/

536 Östman, Ö., Ekblom, B. & Bengtsson, J. (2003) Yield increase attributable to aphid predation by
537 ground-living polyphagous natural enemies in spring barley in Sweden. *Ecological Economics*,
538 **45**, 149–158.

539 Ovenden, G.Y., Swash, A.R. & Smallshire, D. (1998) Agri-environment schemes and their
540 contribution to the conservation of biodiversity in England. *Journal of Applied Ecology*, **35**, 955–
541 960.

542 Plantureux, S., Peeters, A. & McCracken, D. (2005) Biodiversity in intensive grasslands: Effect of
543 management, improvement and challenges. *Agronomy Research*, **3**, 153–164.

544 Pretty, J., Toulmin, C. & Williams, S. (2011) Sustainable intensification in African agriculture.
545 *International Journal of Agricultural Sustainability*, **9**, 5–24.

546 Pywell, R.F., Heard, M.S., Woodcock, B.A., Hinsley, S., Ridding, L., Nowakowski, M. & Bullock, J.M.
547 (2015) Wildlife-friendly farming increases crop yield: evidence for ecological intensification.
548 *Proceedings of the Royal Society of London B: Biological Sciences*, **282**, 20151740.

549 Roberts, D.J. & March, M.D. (2014) Feeding systems for dairy cows: homegrown versus by-product
550 feeds. *Recent Advances in Animal Nutrition 2013* (eds P.C. Garnsworthy & J. Wiseman), pp. 61–

551 69. Context Publications, Packington.

552 Scottish Agricultural College. (2006) *FeedByte*. Scottish Agricultural College, Edinburgh.

553 Simons, N.K. & Weisser, W.W. (2017) Agricultural intensification without biodiversity loss is possible
554 in grassland landscapes. *Nature Ecology & Evolution*, **1**, 1136–1145.

555 Smol, J.P., Wolfe, A.P., Birks, H.J.B., Douglas, M.S. V, Jones, V.J., Korhola, A., Pienitz, R., Rühland, K.,
556 Sorvari, S., Antoniades, D., Brooks, S.J., Fallu, M.-A., Hughes, M., Keatley, B.E., Laing, T.E.,
557 Michelutti, N., Nazarova, L., Nyman, M., Paterson, A.M., Perren, B., Quinlan, R., Rautio, M.,
558 Saulnier-Talbot, E., Siitonen, S., Solovieva, N. & Weckström, J. (2005) Climate-driven regime
559 shifts in the biological communities of arctic lakes. *Proceedings of the National Academy of*
560 *Sciences of the United States of America*, **102**, 4397–402.

561 Symstad, A.J., Siemann, E. & Haarstad, J. (2000) An experimental test of the effect of plant functional
562 group diversity on arthropod diversity. *Oikos*, **89**, 243–253.

563 Tilman, D., Balzer, C., Hill, J. & Befort, B.L. (2011) Global food demand and the sustainable
564 intensification of agriculture. *Proceedings of the National Academy of Sciences of the United*
565 *States of America*, **108**, 20260–4.

566 Toma, L., March, M., Stott, A.W. & Roberts, D.J. (2013) Environmental efficiency of alternative dairy
567 systems: A productive efficiency approach. *Journal of Dairy Science*, **96**, 7014–7031.

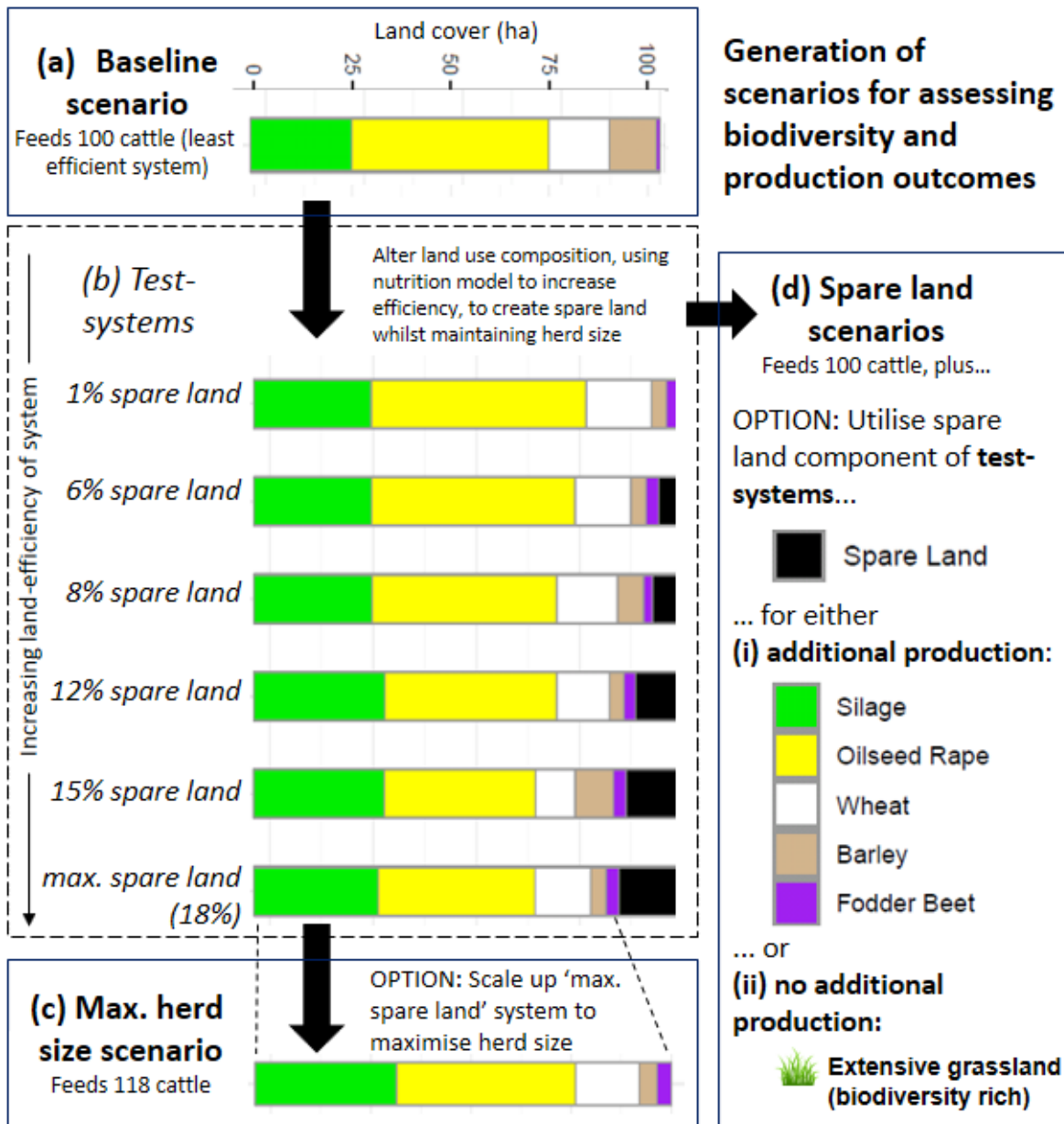
568 Tschardtke, T., Clough, Y., Wanger, T.C., Jackson, L., Motzke, I., Perfecto, I., Vandermeer, J. &
569 Whitbread, A. (2012) Global food security, biodiversity conservation and the future of
570 agricultural intensification. *Biological Conservation*, **151**, 53–59.

571 Valin, H., Sands, R.D., Van Der Mensbrugge, D., Nelson, G.C., Ahammad, H., Blanc, E., Bodirsky, B.,
572 Fujimori, S., Hasegawa, T., Havlik, P., Heyhoe, E., Kyle, P., Mason-D’croz, D., Paltsev, S., Rolinski,
573 S., Tabeau, A., Van Meijl, H., Von Lampe, M. & Willenbockel, D. (2014) The future of food
574 demand: understanding differences in global economic models. *Agricultural Economics*, **45**, 1–
575 17.

576 White, P.J.C., Lee, M.A., Roberts, D.J. & Cole, L.J. (2019) Data from: Routes to achieving sustainable
577 intensification in simulated dairy farms - the importance of production efficiency and
578 complimentary land uses. Edinburgh Napier University Repository.
579 <https://doi.org/10.17869/enu.2019.1497843>

580 Wilson, W., Abernethy, V., Murphy, K., Adam, A., McCracken, D., Downie, I., Foster, G., Furness, R.,
581 Waterhouse, A. & Ribera, I. (2003) Prediction of plant diversity response to land-use change on
582 Scottish agricultural land. *Agriculture, Ecosystems & Environment*, **94**, 249–263.

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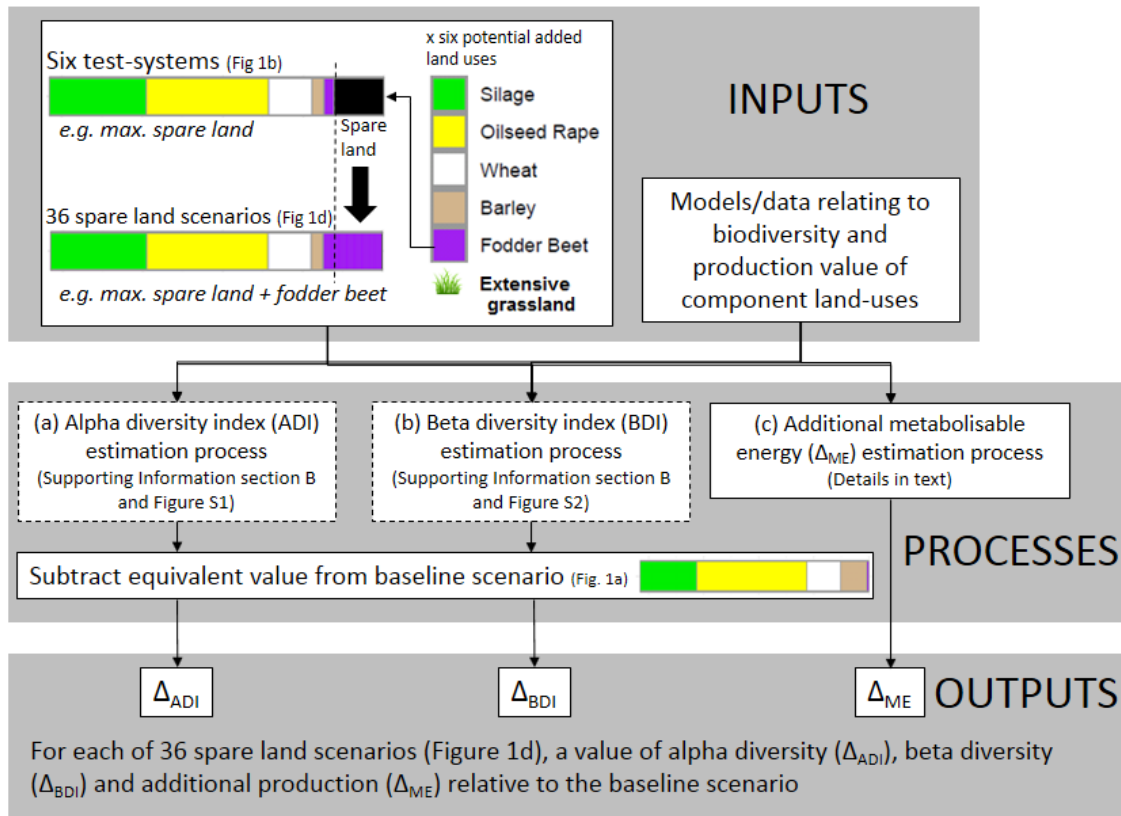
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FIGURE 1. The generation of 38 dairy system scenarios. The feed ration requiring the most land (103.8 ha) was designated the baseline scenario. Altering the farm's crop composition generated spare land (b), without reducing herd size or milk yield (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).

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Further information is provided in the text. More detailed information on the estimation of (a)

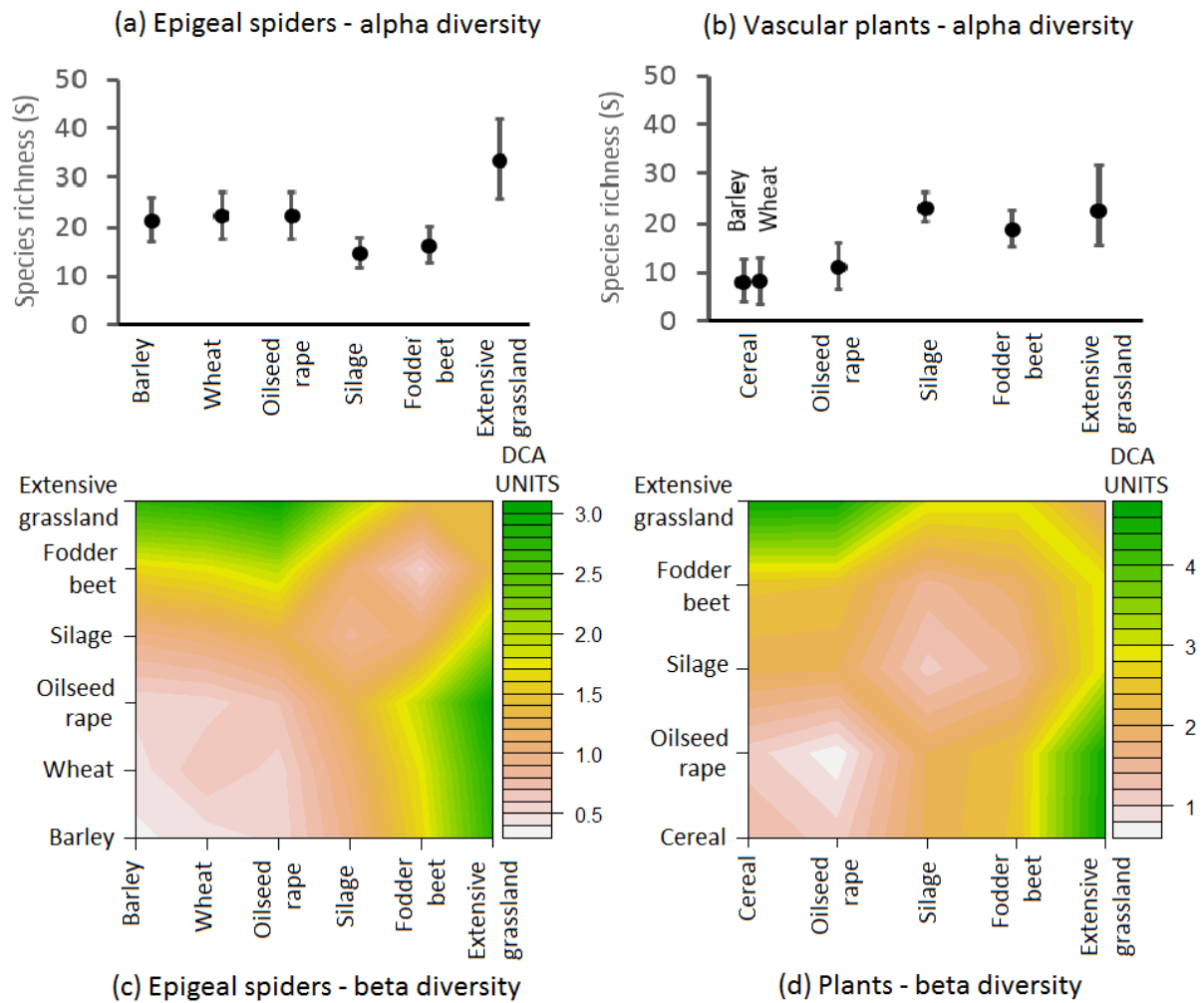
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alpha-diversity and (b) beta-diversity indices (shown in dashed boxes), with sub-workflows and

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model code, are provided in Appendices S2-S4.

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FIGURE 3. Alpha and beta-diversity estimates for plants and epigeal spiders for each land-use from

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datasets in Downie *et al.* (1999) and Wilson *et al.* (2003): (a)-(b) estimated field-scale species

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richness (bars representing upper and lower 95% confidence limits); (c)-(d) contour plots

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representing mean detrended correspondence analysis (DCA) distances between and within land-

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uses (a measure of beta-diversity). Land-uses are ordered to minimise DCA distances across the

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primary and secondary diagonals in (c)-(d).

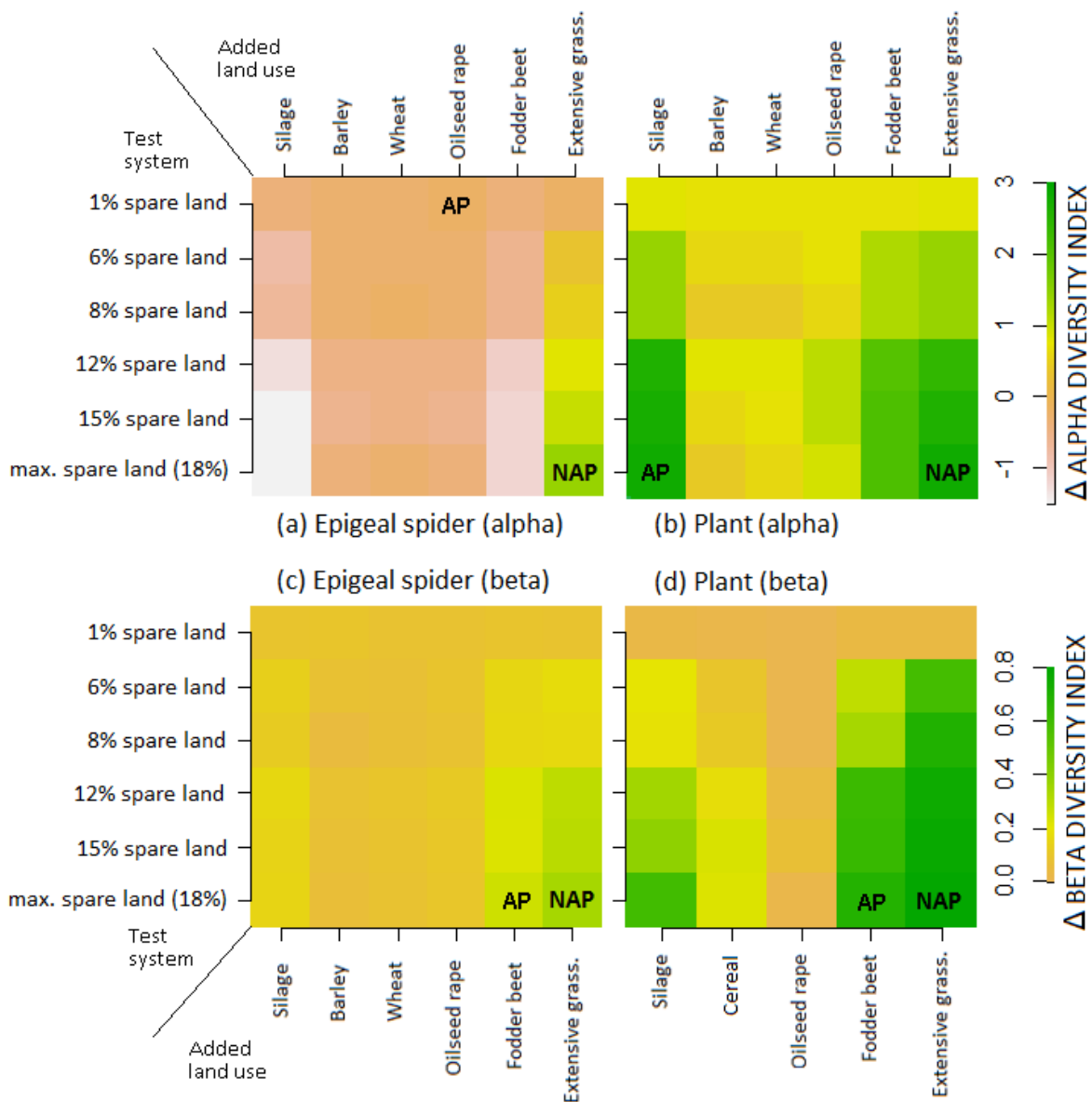
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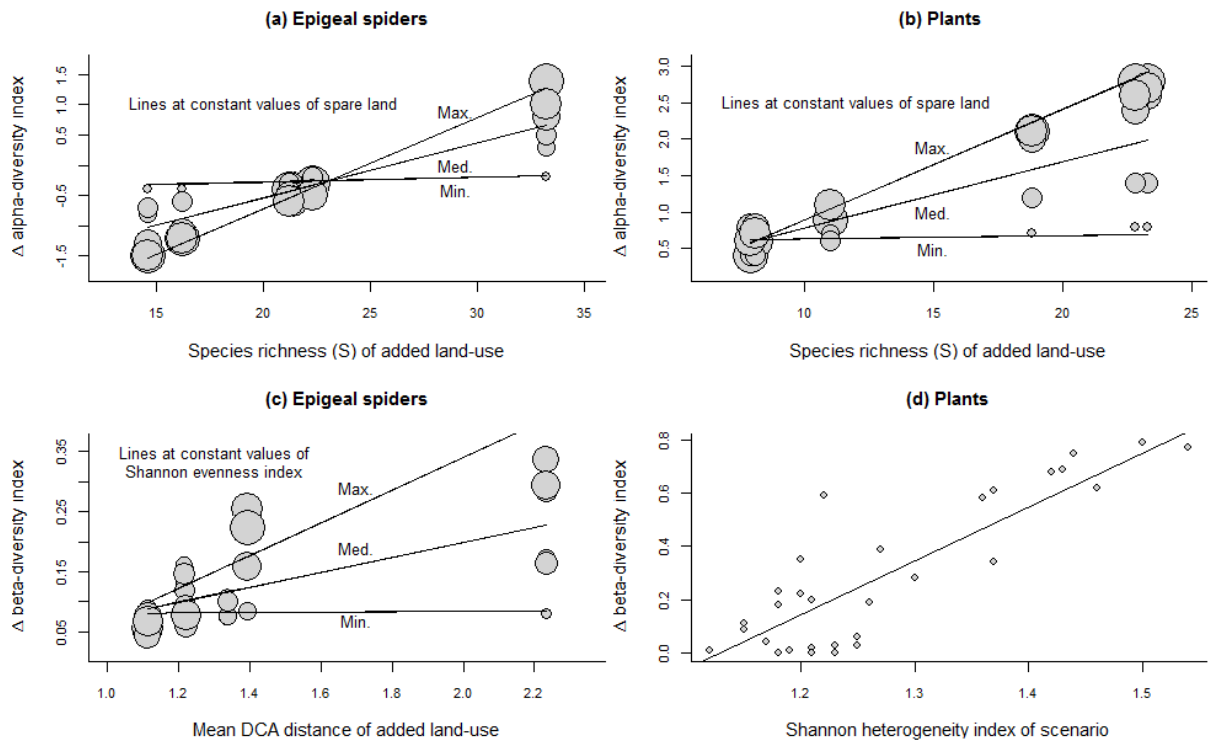
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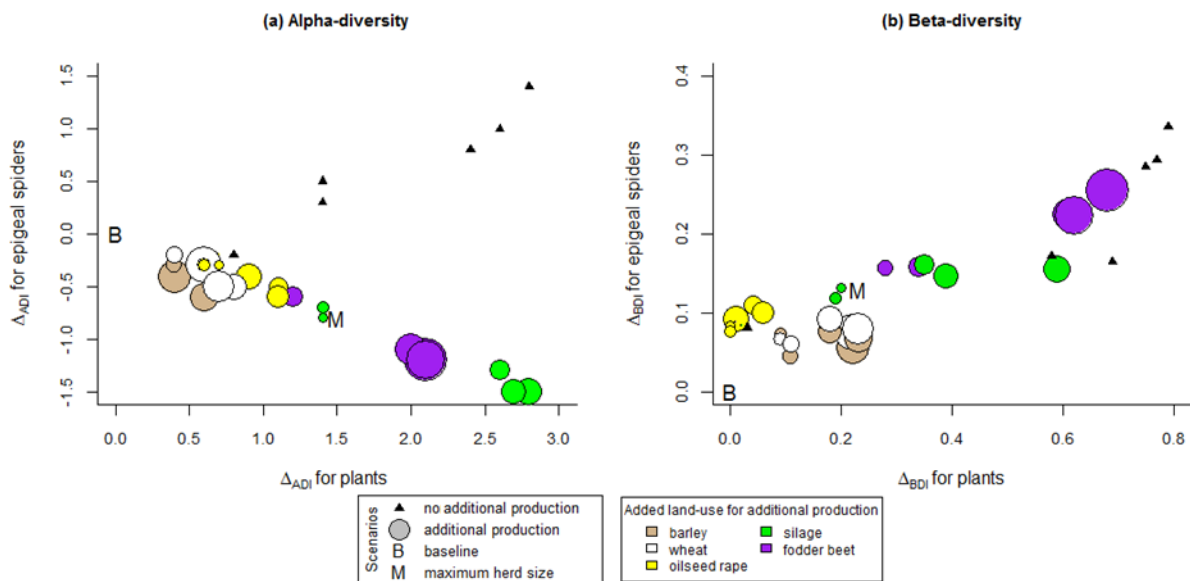


611 **FIGURE 4.** The (a)-(b) change in alpha-diversity index (Δ_{ADI}) and (c)-(d) change in beta-diversity
 612 index (Δ_{BDI}) for 36 spare-land scenarios relative to the baseline scenario, for spiders and plants.
 613 Each spare-land scenario is defined by a test-system (y-axis) and a land-use that is added (x-axis)
 614 to the spare land component of that test-system (Fig. 1). For each plot, the scenario marked 'AP'
 615 represents the highest value for an 'additional production' scenario, and 'NAP' represents the
 616 highest value for a 'no additional production' scenario (Fig. 1d). Derivation of alpha- and beta-
 617 diversity indices from field-scale biodiversity data are described in the text.
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620
 621 **FIGURE 5. Plots of the best AIC_c models presented in Table 2. For presentation purposes, on (a)-(c)**
 622 **circle size is indicative of the relative value on a z-axis (in [a] and [b] this represents the quantity of**
 623 **spare land available, and in [c] this represents the natural logarithm of the Shannon land-use**
 624 **evenness index of the scenario). Interactions on (a)-(c) have been indicated by selecting three**
 625 **constant values on the z-axis (the minimum, median and maximum), and showing the cross-**
 626 **section of the modelled plane at that value. Derivation of alpha- and beta-diversity indices from**
 627 **field-scale biodiversity data are described in the text.**

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637 **FIGURE 6. Plots of diversity indices, (a) Δ_{ADI} and (b) Δ_{BDI} , for spiders and plants for spare-land**
638 **scenarios (Fig. 1d). For ‘additional production’ scenarios, the estimated additional metabolisable**
639 **energy (ME) is indicated by the relative radius of the circle (max for both plots is 1,783 GJ dry**
640 **matter). The ‘no additional production’ scenarios (addition of extensive grassland to spare land)**
641 **do not have additional ME so are shown by \blacktriangle . For comparison, the baseline scenario (Fig. 1a) and**
642 **max. herd size scenario (Fig. 1c) are shown by ‘B’ and ‘M’ respectively. Derivation of alpha- and**
643 **beta-diversity indices from field-scale biodiversity data are described in the text.**

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

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		In model set	
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

676 **Table 2. Ranking table for models of changes in alpha- and beta-diversity from the baseline**
677 **scenario, with the number of model parameters (k), small-sample AIC (AICc), difference in AICc**
678 **from the lowest AICc value ($\Delta AICc$) and the Akaike's weight (w_i) of each. Model terms are defined**
679 **in Table 1. [NULL] = null model. The form $x*z$ indicates an interaction between x and z . The**
680 **confidence set (summed $w_i \geq 0.90$) is in bold. For brevity, only the confidence set, the model**
681 **immediately outside it, and the null model are displayed. The sample size for each model was 36,**
682 **representing the 36 spare land scenarios. Parameter estimates and SEs for all confidence set**
683 **models are in Supporting Information Table S2.**

	Epigeal spiders					Plants				
DIVERSITY	MODEL	k	AICc	$\Delta AICc$	w_i	MODEL	k	AICc	$\Delta AICc$	w_i
Alpha (Δ_{ADI})	RICH*SPARE	4	-50.4	0.0	>0.99	RICH*SPARE	4	-46.3	0.0	>0.99
	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 (Δ_{BDI})	DCA*EVEN	4	-117.3	0.0	0.87	HETER	2	-25.8	0.0	0.71
	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...					

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