1   **Small-scale agricultural landscapes and organic management support wild bee**

2   **communities of cereal field boundaries**

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4   Anne-Kathrin Happe[1,2†], Friederike Riesch[1,3†], Verena Rösch[1,4], Róbert Gallé[1,5], Teja

5   Tscharntke[1] and Péter Batáry[1,6*]

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7   [1]Agroecology, University of Goettingen, Grisebachstr. 6, 37077 Göttingen, Germany;

8   [2]Ecological Networks, Technische Universität Darmstadt, Schnittspahnstr. 3, 64287

9   Darmstadt, Germany; [3]Grassland Science, University of Goettingen, von-Siebold-Str. 8,

10   37075 Göttingen, Germany; [4]Institute for Environmental Sciences, University of Koblenz –

11   Landau, Fortstr. 7, 76829 Landau, Germany; [5]Department of Ecology, University of Szeged,

12   Közép fasor 52, 6726 Szeged, Hungary; [6]GINOP Sustainable Ecosystems Group, MTA

13   Centre for Ecological Research, Klebelsberg Kuno u. 3, 8237 Tihany, Hungary.

14   [†]Equally contributing authors.

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16   *Corresponding author: Grisebachstr. 6, 37077 Göttingen, Germany. Tel.: +49-551-3922358.

17   Fax: +49-551-398806. E-mail: pbatary@gmail.com (P. Batáry).

18 **Abstract**

19 Pollinators are sensitive to agricultural intensification at both local and landscape scales. High

20 configurational landscape heterogeneity due to small fields and a high amount of field

21 boundaries is hypothesized to enhance farmland biodiversity. Here, we investigated the

22 effects of organic vs. conventional farming in large- vs. small-scale agricultural landscapes on

23 wild bee communities and their floral resources to improve conservation schemes for

24 pollinators. We sampled bees in Central Germany at the field boundaries of 18 pairs of

25 conventionally and organically managed winter wheat fields along the former iron curtain

26 using pan traps and trap nests. Around traps, we surveyed species richness and flower cover

27 of insect-pollinated forbs. Compared to conventional farming, organic farming was related to

28 higher insect-pollinated forb species richness and flower cover in the field boundaries,

29 presumably due to the lack of herbicide use. Interestingly, small-scale agriculture did not

30 counteract the loss of floral resources under conventional management, as the difference to

31 organic management was even larger in Western small-scale agriculture. Organic farming, but

32 not small-scale agriculture, enhanced species richness of solitary bees, which is in line with

33 their small home ranges. In contrast, bumblebees benefitted only from small-scale agriculture,

34 which matches with their high dispersal ability. Despite similar levels of abundance and

35 diversity of trap-nesting bees in the two landscape types, brood cell parasitism was also

36 higher in small-scale agriculture. Both organic farming and small-scale agriculture directly

37 and indirectly supported different groups of wild bees, suggesting long-term benefits for

38 conservation. Agri-environment schemes should acknowledge the so far neglected benefits of

39 small-scale agriculture for biodiversity and its potential services.

40

41 **Keywords:** Agri-environment schemes, insect-pollinated plants, landscape configuration, pan

42 trap, pollinators, trap nest

43

1. Introduction

Wild bees are strongly affected by agricultural intensification and landscape homogenization. At the local scale, a major threat is the increased input of pesticides and fertilizers (Robinson and Sutherland, 2002), whereas changes in the agricultural landscape due to increased field sizes and the loss of suitable habitats exert pressure on biodiversity and ecosystem services (Steffan-Dewenter et al., 2002; Tscharntke et al., 2005; Rundlöf et al., 2008; Bommarco et al., 2010). Declines in the abundance and diversity of native bees, and their services can be explained by the isolation from critical floral and nesting resources (Steffan-Dewenter and Tscharntke, 1999; Kremen et al., 2002). Despite the importance of wild bees providing pollination services and the increasing awareness of this problem (Klein et al., 2007; Aizen et al., 2009; Garibaldi et al., 2013), pollinators in agricultural landscapes are still at risk.

Agri-environment schemes (AES) may be powerful mitigation tools to enhance habitat quality for pollinators and combat biodiversity loss in intensively cultivated areas (Batáry et al., 2015). Instead of establishing new protection areas, these schemes integrate conservation efforts into the agricultural landscape. Farmers may provide flower strips, nesting resources and restore semi-natural and natural areas adjacent to crops (Garibaldi et al., 2014, 2016; Kleijn et al., 2015).

Even though AES are often considered as an expensive and complex form of environmental protection (Batáry et al., 2015), the investment is observed to pay off. AES enhance ecosystem services such as pollination and natural pest control by increasing farmland heterogeneity and extensive farming practices (Inclán et al., 2015). Among AES measures, organic farming is one of the most well-established management approaches (Batáry et al., 2013). The high value of organic management for agro-biodiversity has been shown for several agricultural systems (e.g. Schmidt et al., 2005; Rundlöf et al., 2008; Schneider et al., 2014). Including data from 30 years and different farming systems in a meta-

69   analysis, Tuck et al. (2014) could show for example that organic farming increased species

70   richness by 30% with the strongest positive effects in cereals and mixed crops. Regarding

71   functional groups they emphasized the positive effect of organic farming on plants and

72   pollinators.

73   Various studies have shown that the composition of the surrounding landscape

74   moderates the effectiveness of AES in general, and organic farming in particular (Rundlöf et

75   al., 2007; Batáry et al., 2011). For example, pollinators profit from agri-environmental

76   management in simple but not in complex landscapes (Batáry et al., 2011). Landscape

77   configuration is an important factor affecting biodiversity and ecosystem services, but little is

78   known about how landscape configurational heterogeneity shapes biodiversity patterns under

79   AES (Belfrage et al., 2005; Marshall et al., 2006; Concepción et al., 2012). The shift from

80   small-scale family farms to large-scale agriculture has severely decreased landscape

81   configurational heterogeneity. This process is accompanied by a trend towards specialization

82   and the increase of field sizes on the expense of mixed farming and high quality edges, such

83   as hedgerows (Robinson and Sutherland, 2002; Holzschuh et al., 2010).

84   In this study, we used a novel approach to disentangle the interlinked effects of local

85   management and landscape configuration on wild bees. At the former inner German border

86   (iron curtain), two opposed farming strategies have led to diverging field size patterns. While

87   small private farms shape West Germany's agriculture, Eastern farms were merged and

88   organised in large cooperatives. In 1989, average farm size in the West was 18.17 ha. In East

89   Germany, farms were on average 250 times as large due to collectivisation under planned

90   economy (Koester, 1999). Similarly, in many Central and Eastern European countries,

91   collectivised family farms were merged to large industrial farms of several hundred or

92   thousand hectares in size (Davidova et al., 2012; Sutcliffe et al., 2015). Today, post-

93   communist legacy effects in the agricultural landscape are still visible and affect farmland

94   biodiversity (Báldi and Batáry, 2011; Konvicka et al., 2016; Batáry et al., 2017). We studied

95   the effects of farming practice and landscape configuration based on these historically grown

96   structures alongside the former iron curtain. Our research focused on wild bees, which

97   contribute crucially to ecosystem service provision in agricultural landscapes, and on insect-

98   pollinated forbs, which offer essential food resources for this prominent group of pollinators.

99   Both bees and forbs are highly sensitive indicators for agricultural intensification at the local

100   and the landscape scale, and thus belong to the major target groups of AES (Batáry et al.,

101   2015; Garibaldi et al., 2013; Kennedy et al., 2013). We expected local management and

102   landscape configuration to strongly affect abundance and species richness of both wild bees

103   and their floral resources in wheat field boundaries. We addressed the following research

104   questions: (1) Do insect-pollinated forb and wild bee species richness and abundance in cereal

105   field boundaries increase in small-scale compared to large-scale agricultural landscapes? (2)

106   Does organic farming enhance forb species richness and cover and pollinator diversity and

107   abundance? (3) Can we detect a coupled effect of farming practice and landscape

108   configuration on wild bees and their floral resources?

109

110   **2. Materials and methods**

111   *2.1. Study sites and study design*

112   We sampled a total of 36 winter wheat fields in southern Lower Saxony (former democratic

113   West Germany, hereafter 'West') and northern Thuringia (former communist German

114   Democratic Republic, hereafter 'East') in 2013, at both sides of the former inner German

115   border (Appendix S1 in Supplementary Material). Intensive cropland (mainly cereals and oil

116   seed rape) and pastures dominated the surrounding landscape accompanied by semi-natural

117   habitats such as forest patches and hedgerows. Annual precipitation was 581 mm and mean

118   annual temperature was 8.9 °C (measuring station Mühlhausen, 190 m a.s.l.) in Northern

119   Thuringia (DWD 2013). Western fields were situated around the city of Göttingen (weather

120   measuring station at 176 m a.s.l.) with a mean annual precipitation of 651 mm and a mean

121   annual temperature of 9.2 °C (DWD 2013). Edge length and average field size recorded in a

122   500 m buffer around study fields provided evidence for the strong contrast in landscape

123   configuration between large-scale East and small-scale West. Field edges in the West were

124   1.7 times longer and fields six times smaller than in the East), whereas the proportion of

125   agricultural area was similar in both landscape types (Table 1). The percentage of organic

126   farmland in the study regions accounted for 2.8% of the agricultural area in Lower Saxony

127   and 4.2% in Thuringia in 2015 (BMEL, 2015). We selected 18 pairs of organic and

128   conventional fields in both regions for simultaneous testing of the influence of landscape

129   configuration (large-scale East vs. small-scale West) and farming practice. In both regions,

130   conventional and organic systems were clearly differentiated based on management intensity

131   (no application of synthetic pesticides in organic farming; amount of nitrogen fertilizer used

132   about five times lower, and yields half as high in organic compared to conventional farming

133   (Batáry et al., 2017)). Fields belonging to one pair were located within the area of one village

134   in the immediate vicinity of each other (distance between paired fields in East: 2.60 ± 0.58

135   km; in West: 1.10 ± 0.22 km). Since organic farms are not as widely distributed in the East as

136   in the West, we had to apply a partly cross-nested study design (Batáry et al., 2017): In three

137   villages (in both East and West), we selected two pairs of fields belonging to the same

138   farmers, resulting in three villages with one pair and three villages with two pairs of organic

139   and conventional fields per landscape type (altogether 24 farms). On average, study field size

140   was seven times larger in the East than in the West (Table 1). Along each field, we selected

141   one grassy field boundary with homogeneous cover (excluding sown flower mixtures or

142   hedgerows) for pollinator sampling and vegetation surveys.

143

2.2. Vegetation surveys

Detailed vegetation surveys were conducted once in mid-June 2013, following the sampling protocol used by Batáry et al. (2013). We assessed species richness and cover of insect-pollinated forbs in three botanical plots of 5×1 m size, located in the centre of the grassy field boundary and separated by 10 m from each other. Additionally, percent cover of flowering plants, which might attract pollinators, was recorded in the plots around pan traps (see below).

2.3. Pollinator sampling

In each field boundary, we sampled pollinators using three yellow pan traps placed on sticks of 1 m height. We established the traps parallel to the field edge in the centre of the vegetation survey plots at the field boundary. Yellow traps are known to be most attractive for wild bees and are frequently used to compare pollinator diversity between different study sites (Grundel et al., 2011; but see Morandin and Kremen, 2013). We opened the traps for seven days at each site during three survey periods (May 6-16, May 28 - June 6 and June 24 - July 4, 2013). Trapped pollinators were stored in 70% ethanol, dried, needled and identified to species level.

      We sampled trap nesting bees using two trap nests per field boundary consisting of plastic tubes filled with reed internodes (Gathmann et al., 1994). Each trap nest was composed of two trapping cylinders set up on two wooden posts in 15 m distance from each other in spring (April 22-24, 2014). We collected the traps in autumn (September 13-16, 2014). We opened occupied reed nests in the laboratory, examined them for parasitism and identified trap-nesting bees and wasps to genus level.

2.4. Statistical analyses

168   We used the software R 3.3.2 (R Core Team, 2016) to conduct statistical analyses. We tested

169   the effects of farmland management and landscape configuration on insect-pollinated forb

170   diversity and cover, wild bee species richness and abundance (separately for solitary bees and

171   bumblebees) and genus richness, cell number and parasitized cell number of trap-nesting bees

172   and wasps by general and generalised linear mixed effect models (GLMM) using the

173   functions 'lmer' and 'glmer' of the package 'lme4' version 1.1-12 (Bates et al., 2015). To

174   avoid spatial and temporal autocorrelation, we pooled all samples per field by summing up

175   insect numbers. Based on the strong contrasts in landscape configuration and management

176   intensity (Table 1) we employed the factors 'landscape scale' (East: large-scale vs. West:

177   small-scale) and farm 'management' (organic vs. conventional) and their interaction as

178   categorical predictor variables. As some fields of the same management type belonged to the

179   same farmer, we included nested random effects, with 'farmer' nested in 'village' and 'pair'

180   nested in 'village' (Batáry et al., 2017). In case of overdispersion, we accounted for

181   individual-level variability by including 'field' as additional random factor. We used visual

182   methods (quantile-quantile plots and homogeneity plots of residuals vs. fitted values) and

183   Shapiro-Wilk-tests to check the distribution of the residuals. Models were fitted with normal,

184   Poisson, or negative binomial distribution according to the error distribution of the response

185   variable (Table 2). We calculated all models nested in the global model by the command

186   'dredge' in the package 'MuMin' version 1.16.4 (Barton, 2016) and compared them based on

187   Akaike Information Criterion corrected for small sample size (AICc). We performed model

188   averaging (Anderson and Burnham, 2002) if the top model and subsequent models differed

189   less than two units in AICc. Model-averaged parameter estimates were calculated over the

190   subset of models including the parameter (conditional average) to avoid shrinkage towards

191   zero (Grueber et al., 2011). We present the 95% confidence interval (CI) of parameter

192   estimates and the relative importance of each parameter. Relative importance is 0%, when the

8

193   parameter does not appear in the top model set, and reaches 100%, when the parameter is

194   present in all top models. Figures represent mean and standard error of the mean (SEM) of

195   species or genus richness and abundance, respectively, for each landscape and management

196   type.

197

198   **3. Results**

199   *3.1. Insect-pollinated forbs*

200   In total, we recorded 175 plant species in the cereal field boundaries. According to Klotz et al.

201   (2002), 70 of these species were insect-pollinated forbs (Table S1). Species richness of insect-

202   pollinated forbs was highest in organic field boundaries in both large-scale agriculture (43

203   species) and small-scale agriculture (47 species), whereas only 15 species were found in

204   conventional field boundaries in small-scale agriculture, compared to 37 species in large-scale

205   agriculture. For insect-pollinated forb species richness we retained the full model including

206   landscape scale, management, and the interaction term as single top model (ΔAICc < 2).

207   There was an interaction between landscape scale and management, indicating a stronger

208   management effect on insect-pollinated forb species richness in small-scale agriculture than in

209   large-scale agriculture (Table 2, Fig. 1a). Forb cover differed between management types

210   (higher cover in organic than in conventional field boundaries), but not between landscape

211   scales, as the single best candidate model for insect-pollinated forb cover contained only

212   management as predictor variable (Fig. 1b).

213

214   *3.2.Bumblebees and solitary bees*

215   We collected a total of 1915 wild bees (1512 solitary and 403 social bees) using pan traps and

216   identified 81 species belonging to 16 genera (Table S2). The highest species richness and

217   abundance occurred within the taxon *Andrena* with the species *A. nigroaenea* and *A.*

218   *haemorrhoa* being the most frequent. Individuals of the managed European honeybee (*Apis*

219   *mellifera*) were excluded from the analyses.

220   Species richness and abundance of social wild bees, i.e. bumblebees, in field boundaries

221   depended on the landscape type (Table 2, Fig. 2a,c). However, the landscape effect on species

222   richness was less evident than on abundance, which was more than two times higher in small-

223   scale agriculture relative to large-scale agriculture. In contrast, species richness and

224   abundance of solitary bees were affected by management, but not by landscape scale (Fig.

225   2b,d). Again, the effect on abundance was stronger than on species richness, with the number

226   of solitary bees being reduced almost by one third under conventional compared to organic

227   management.

228

229   *3.3. Trap nesting bees and brood cell parasitism*

230   We found 13 genera of trap nesting bees and wasps checking 7126 brood cells. Overall,

231   3.21% of cells were parasitized (Table S3). Management was the only factor explaining genus

232   richness and cell number of trap nesting bees and wasps, but evidence for a positive effect of

233   organic management was limited (Table 2, Fig. 3a,b). However, there were about twice as

234   many parasitized cells in small-scale agriculture than in large-scale agriculture, and the single

235   best candidate model for the number of parasitized cells strongly supported an effect of

236   landscape scale (Fig. 3c).

237

238   **4. Discussion**

239   Based on differences in landscape configuration derived from opposing agricultural land-use

240   history in former East and West Germany, our study allowed disentangling landscape-scale

241   (large-scale vs. small-scale agriculture) and local (organic vs. conventional management)

242   effects on wild bee communities and their floral resources. Organic management consistently

243   increased the cover and species richness of insect-pollinated forbs in the cereal field

244   boundaries, in both large-scale and small-scale agricultural landscapes. However, the more

245   pronounced increase in forb species richness in small-scale agriculture compared to large-

246   scale agriculture suggested a higher effectiveness of organic management in small-scale

247   systems. The positive effect of organic management on forbs was accompanied by an

248   enhanced abundance of solitary wild bees. By contrast, bumblebee abundance did not increase

249   under organic management, whereas it was much higher in small-scale than in large-scale

250   systems. Similarly, parasitism of trap nesting bees and wasps was higher in the small-scale

251   than in the large-scale agriculture.

252

253   *4.1. Insect-pollinated forbs*

254   Forb species richness in field boundaries was notably reduced under conventional

255   management, particularly in small-scale agriculture. The decrease in diversity and cover of

256   forbs observed in landscapes characterized by high cover of conventionally managed arable

257   land has been explained by the intensive use of herbicides and the loss of semi-natural refuge

258   habitats (Roschewitz et al., 2005; Gaba et al., 2010; Dainese et al., 2016). Forb species

259   richness in field boundaries might also decrease with increasing fertilizer application in the

260   adjacent conventional crop field (Kleijn et al., 2009). In our study, there was no evidence that

261   conventional farmers applied higher amounts of fertilizers and pesticides in small-scale

262   compared to large-scale agriculture (Batáry et al. 2017). Given the similar level of agro-

263   chemical applications in both regions, we expected that higher edge lengths in small-scale

264   agriculture counteract habitat loss and enhance forb species richness and cover in

265   conventional field boundaries. In contrast, we found the difference in species richness and

266   cover of forbs between organic and conventional management to be much more pronounced

267   in small-scale than in large-scale agricultural landscapes. Consequently, in our study,

268 heterogeneous small-scale agriculture did not reduce the effectiveness of organic farming in

269 enhancing biodiversity, as suggested by other authors (Tscharntke et al., 2012). Based on our

270 experiences a tentative explanation could be that conventional field margins in the large-scale

271 agricultural landscapes in the East are less frequently managed by mowing compared to the

272 small-scale landscapes in the West (Chaudron et al., 2016).

273   Our findings substantiate strong benefits of organic farming for plant species richness

274 and cover, which are well-known and can be explained by the lack of herbicide application

275 (Roschewitz et al., 2005; Rundlöf et al., 2009; Tuck et al., 2014). Although the effectiveness

276 of organic farming has been questioned, for example in comparison to the restoration of semi-

277 natural habitats (Batáry et al., 2015), organic farming has generally been given credit for

278 balancing between multiple sustainability goals, such as productivity, environmental impact,

279 economic profit and human well-being (Reganold and Wachter, 2016).

280   It has been shown that the favourable effect of organic farming in terms of biodiversity

281 conservation increases in areas with high crop cover (Tuck et al., 2014; Batáry et al., 2015).

282 In our comparison of large-scale and small-scale farming systems, however, the proportion of

283 agricultural area covering the surroundings of our study sites was similar in both landscape

284 types (ca. 80%, Table 1). Thus we can largely exclude that differences in plant species

285 richness in organic field boundaries between small- and large-scale agriculture were related to

286 crop cover in the surrounding landscape.

287

288 *4.2. Bumblebees and solitary bees*

289   Our results provide evidence that organic farming succeeds not only in promoting

290 insect-pollinated forbs, but also solitary bee abundances in the wheat field boundaries. Social

291 bees, i.e. bumblebees, did not benefit from organic management but from small-scale

292 agriculture, indicating that bumblebees are more sensitive to landscape-scale intensification

293 than solitary bees. Larger body sizes in social bees translate to larger foraging ranges

294 (Westphal et al., 2006; Greenleaf et al., 2007; Kennedy et al., 2013), suggesting that social

295 bees perceive landscapes at a broader spatial scale than solitary bees (Steffan-Dewenter et al.,

296 2002). Steffan-Dewenter and Tscharntke (1999), who tested the effect of habitat isolation on

297 wild bees, confirmed a positive correlation of body size and foraging distance. In our study,

298 the large foraging ranges of bumblebees, especially of the most common species *Bombus*

299 *terrestris*, may partly explain the positive effect of landscape heterogeneity, i.e. smaller fields

300 with higher boundary lengths per unit area in small-scale agriculture, on bumblebee species

301 richness and abundance. Bumblebees depend strongly on high quality foraging habitat and

302 flower-rich field boundaries (Carvell et al., 2004), which were more available in small-scale

303 agriculture based on higher edge lengths (Table 1). Carvell et al. (2017), for instance, showed

304 that the survival of bumblebee family lineages between years increased significantly, when

305 bumblebees had access to flower rich field boundaries within 250-1,000 m from nesting sites.

306 Our finding that bumblebees benefit from small-scale agricultural landscapes is also in

307 accordance with Morandin et al. (2016), who showed that small-scale restorations of field

308 boundaries with hedgerows could be ecologically and economically cost effective in

309 promoting pollinators in adjacent crops within simplified agricultural landscapes.

310 Kennedy et al. (2013) found only weak effects of landscape configuration on wild bees,

311 but a strong influence of organic management. Contrary to our results, they reported that both

312 solitary and social bees profited from improved habitat conditions under organic management,

313 especially if vegetation diversity in the fields was high. There is no simple explanation why

314 the higher flower-richness we recorded in boundaries of organic fields did not lead to higher

315 bumblebee abundance and/or species richness. We assume that strong landscape-scale effects

316 in our study may have superimposed a potential positive effect of organic farming on

317 bumblebees. It is likely that the higher density of boundaries in small-scale agriculture offers

318   nesting sites and foraging resources at larger spatial scales, reducing the attractiveness of local

319   floral resources provided in organic field margins.

320

321   *4.3. Trap nesting bees and their parasitism*

322   Trap nest communities have been effectively used to study pollinators, predatory wasps, their

323   parasitoids and parasitism, which is an essential ecosystem function regulating host

324   populations (Tylianakis et al., 2006; Pereira-Peixoto et al., 2014, 2016). In our study,

325   parasitism of trap nesting bees and wasps was affected by small-scale agriculture, which

326   enhanced parasitoid abundance independently from farm management and host density.

327   Pereira-Peixoto et al. (2014) found more natural enemies of cavity-nesting bees in

328   heterogeneous habitats (urban-rural interface vs. either urban or rural), which were assumed

329   to provide more resources to parasitoids (Pereira-Peixoto et al., 2016), i.e. floral resources as

330   well as hosts and nesting sites for hosts. Their finding is in accordance with the high

331   parasitoid abundance we detected in small-scale agricultural landscapes, though we did not

332   find more brood cells of hosts or a higher number of cavity nesting bee genera in small-scale

333   agriculture. As opposed to this, Holzschuh et al. (2010) observed that local- and landscape-

334   scale effects on parasitoids were mainly mediated by their hosts, and parasitism rates were

335   marginally affected by local factors. However, in line with our findings, Steckel et al. (2014)

336   reported a similar response of trap nest parasitoids in grasslands at different land use

337   intensities, with parasitoids reacting more sensitive to low configurational landscape

338   heterogeneity than their hosts and perceiving the influence of this factor at larger spatial

339   scales (up to 1500 m). Hence, small-scale landscapes can enhance potential regulation of host

340   populations, which can be generally expected based on constraints in dispersal ability and

341   feeding specialization of natural enemies (Perović et al., 2017). By studying trap-nest

342   communities we were able to show that landscape configuration and management affect host

343 abundance and parasitism differently, implying effects on trophic interactions and ecosystem

344 functions.

345

346 **5. Conclusion**

347 Comparing management and landscape-scale effects on insect-pollinated forbs, bees and their

348 parasitoids, we showed that organic farming and the associated high availability of local floral

349 resources enhanced solitary bees, which are spatially restricted by their small foraging ranges.

350 Perceiving landscapes at a broader spatial scale, bumblebees profited from small-scale

351 agriculture. Therefore, we propose to promote small-scale agriculture, i.e. the reduction of

352 field sizes, as an AES to enhance habitat conditions for wild bee communities in farmland.

353 Next to organic management, the reduction of field sizes may serve as an effective tool to

354 support pollinators in the agricultural landscape.

355

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Figure captions

570

571 **Fig. 1.** Insect-pollinated forb species richness (a) and cover [%] (b) in boundaries of

572 conventionally and organically managed winter wheat fields in small-scale and large-scale

573 agricultural landscapes. Error bars represent SEM (n = 36).

574

575 **Fig. 2.** Bumble bee species richness (a) and abundance (c) and solitary bee species richness

576 (b) and abundance (d) in boundaries of conventionally and organically managed winter wheat

577 fields in small-scale and large-scale agricultural landscapes. Error bars represent SEM (n =

578 36).

579

580 **Fig. 3.** Genus richness (a) and cell number (b) of trap-nesting bees and number of parasitized

581 cells (c) in boundaries of conventionally and organically managed winter wheat fields in

582 small-scale and large-scale agricultural landscapes. Error bars represent SEM (n = 36).

583 **Table 1.** Local management intensity and landscape structure (in 500 m buffer) around study

584 fields (n = 36 fields) in small (West) vs. large (East) scale agricultural systems with organic

585 vs. conventional management (mean ± SEM) during 2013 (based on Batáry et al. 2017)

	Small-scale agriculture (West)		Large-scale agriculture (East)	
	Organic	Conventional	Organic	Conventional
Local management				
Fertilizer (kg N/ha)	21.6 ± 10.9	199.3 ± 4.7	65.3 ± 11.7	193.6 ± 8.6
Pesticides (no. of appl.)	0.0 ± 0.0	4.3 ± 0.3	0.0 ± 0.0	5.2 ± 0.7
Study field size (ha)	3.0 ± 0.5	3.1 ± 0.3	21.8 ± 3.6	20.0 ± 3.0
Surrounding landscape				
Crop cover (%)	73.9 ± 4.1	76.9 ± 4.7	81.0 ± 5.1	85.5 ± 4.5
Edge length (km)	18.3 ± 1.3	19.5 ± 1.2	11.0 ± 0.8	10.8 ± 0.6
Field size (ha)	3.7 ± 0.7	3.3 ± 0.3	21.7 ± 5.5	18.3 ± 2.1

586

587 **Table 2.** Summary table for GLMM results after multimodel averaging of best candidate

588 models showing relative importance of each explanatory variable (Management (M): organic

589 vs. conventional; Landscape scale (L): large-scale vs. small-scale agriculture), its estimated

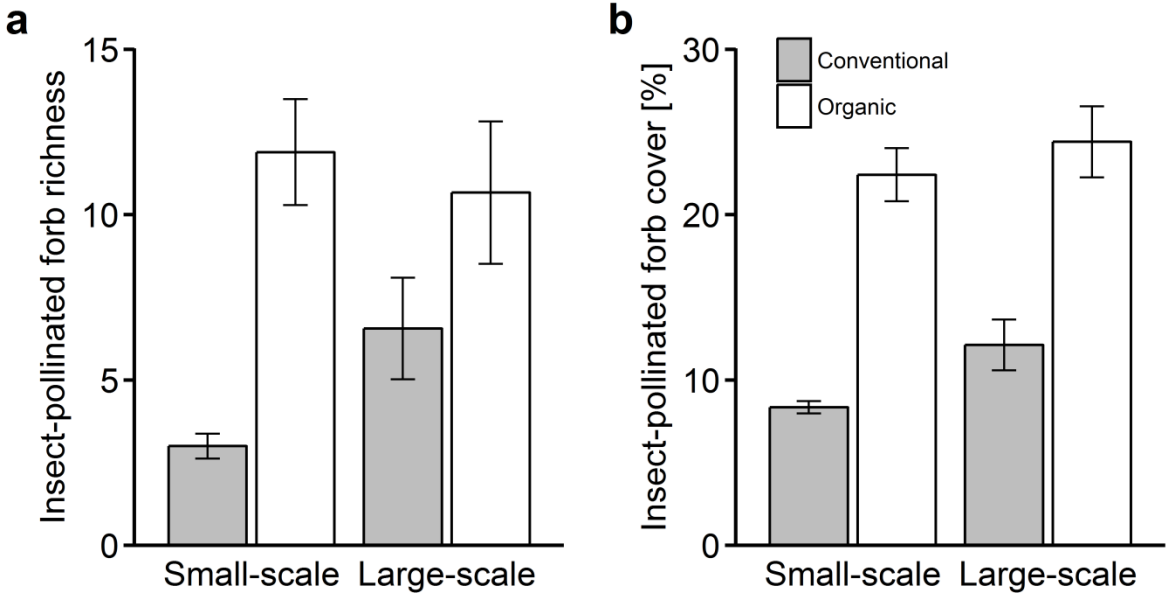590 effect on the response ± 95% CI. Bold values indicate significant effect at $P = 0.05$

Model[*]	Variable	Relative importance (%)[†]	Multimodel estimate ± 95% CI[‡]	
Insect-pollinated forbs				
Species richness[2]	Management	**100**	**0.49**	**± 0.32**
	Landscape scale	**100**	**-0.70**	**± 0.64**
	L × M	**100**	**0.89**	**± 0.54**
Cover[1]	Management	**100**	**0.19**	**± 0.10**
Bumblebees				
Species richness[2]	Landscape scale	66	0.38	± 0.39
Abundance[2]	Landscape scale	**100**	**1.07**	**± 0.58**
Solitary bees				
Species richness[2]	Management	54	0.18	± 0.20
Abundance[3]	Management	**33**	**0.30**	**± 0.04**
Trap nesting bees and wasps				
Genus richness[2]	Management	54	0.31	± 0.36
Cell number[3]	Management	37	0.42	± 0.62
Parasitized cells[3]	Landscape scale	**100**	**1.02**	**± 0.92**

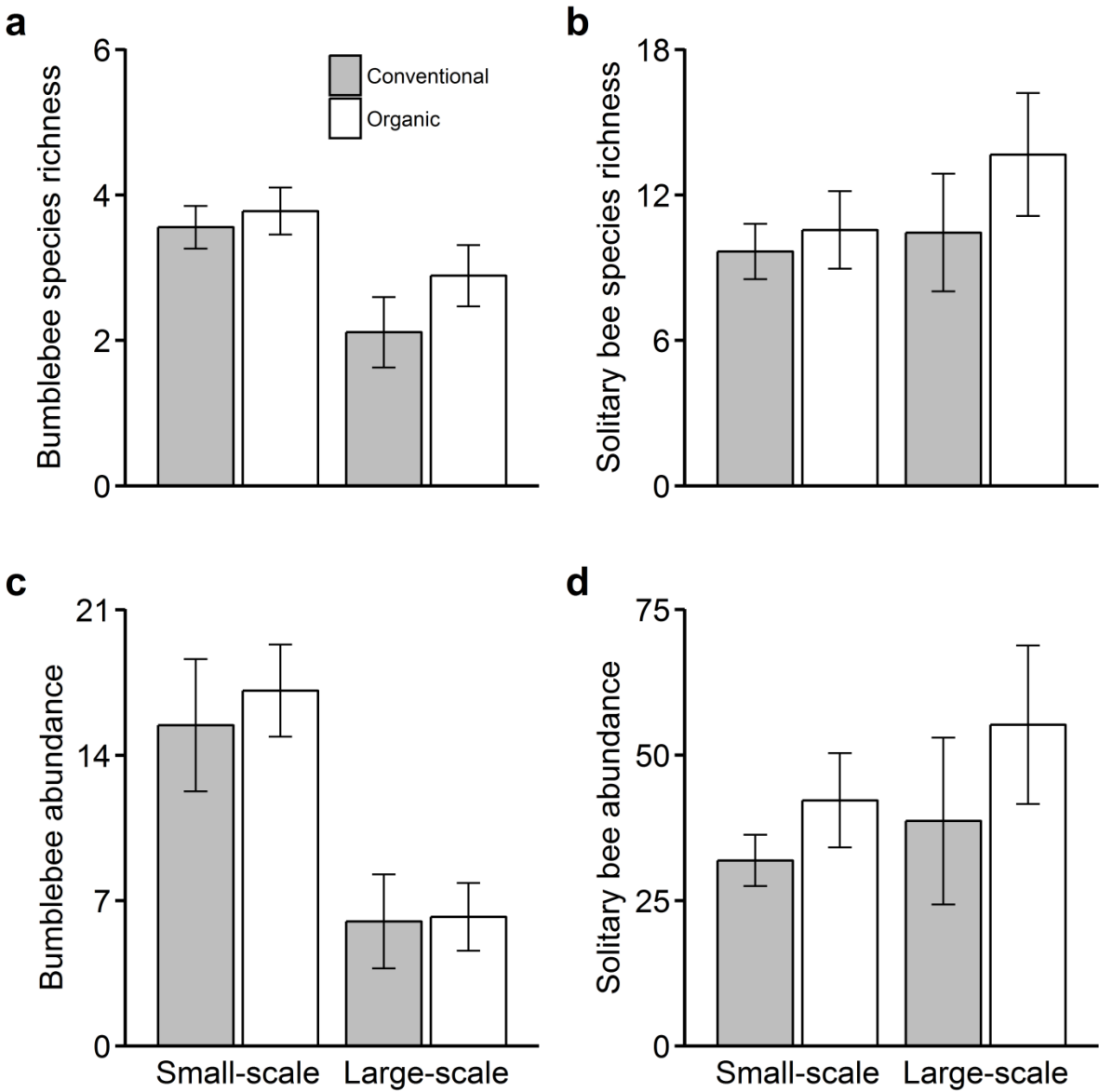[*]Models were fitted with normal (1), Poisson (2), or negative binomial distribution (3).

[†]Each variable's importance within the set of best candidate models (ΔAICc < 2).

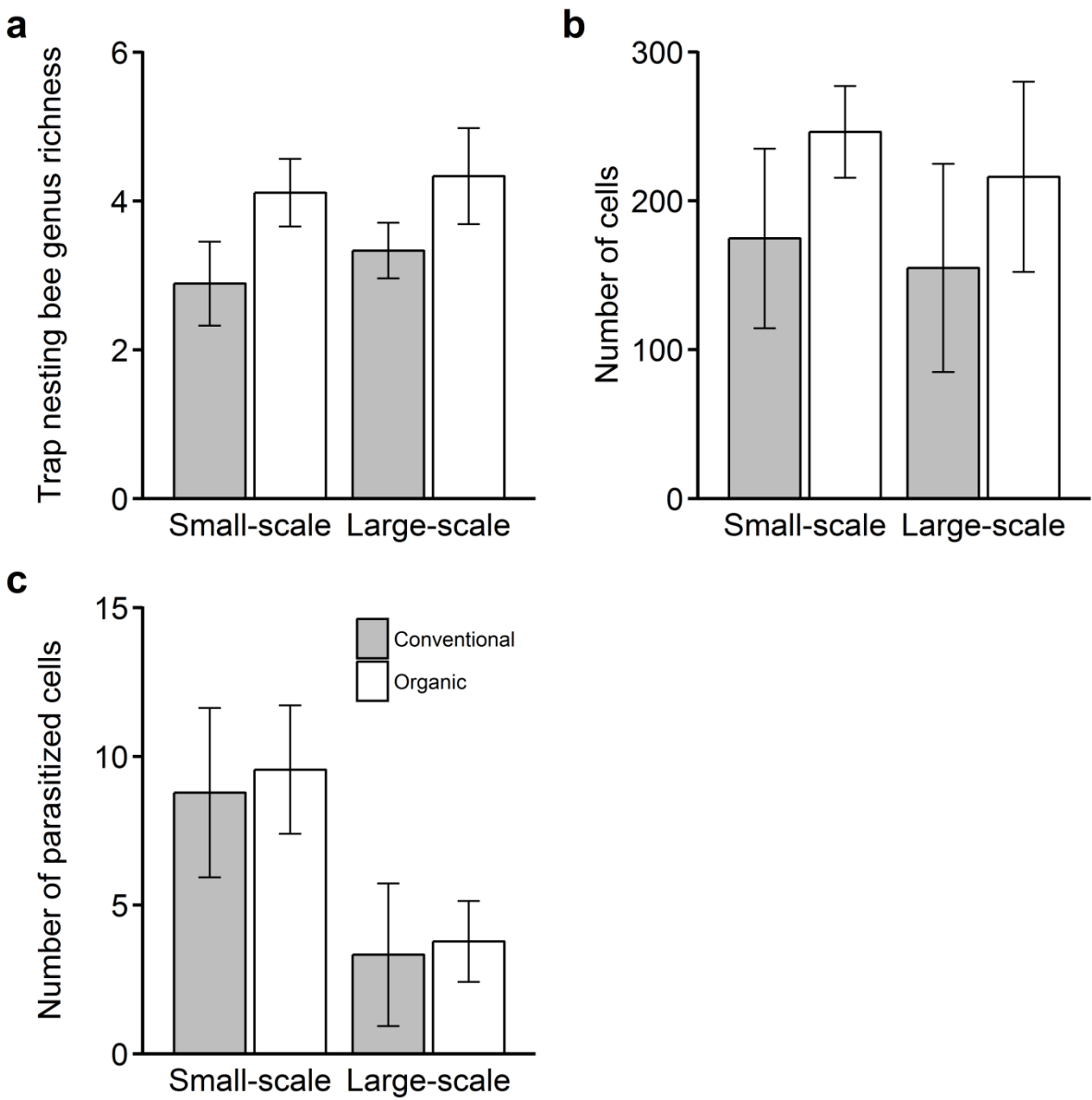[‡]Estimates after multimodel averaging of the best candidate models (ΔAICc < 2).

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592

593   **Fig. 1.**

596 **Fig. 2.**

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599   **Fig. 3.**