**A global synthesis of the effects of diversified farming systems on arthropod diversity within fields and across agricultural landscapes**

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# A global synthesis of the effects of diversified farming systems on arthropod diversity within fields and across agricultural landscapes

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| Keywords: | agricultural management schemes, arthropod diversity, functional groups, landscape complexity, meta-analysis, evenness, biodiversity, organic farming |
| Abstract: | Agricultural intensification is a leading cause of global biodiversity loss, which can reduce the provisioning of ecosystem services in managed ecosystems. Organic farming and plant diversification are farm management schemes that may mitigate potential ecological harm by increasing species richness and boosting related ecosystem services to agroecosystems. What remains unclear is the extent to which farm management schemes affect biodiversity components other than species richness, and whether impacts differ across spatial scales and landscape contexts. Using a global meta-dataset, we quantified the effects of organic farming and plant diversification on abundance, local diversity (communities within fields), and regional diversity (communities across fields) of arthropod pollinators, predators, herbivores, and detritivores. Both organic farming and higher in-field plant diversity enhanced arthropod abundance, particularly for rare taxa. This resulted in increased richness but decreased evenness. While these responses were stronger at local relative to regional scales, richness and abundance increased at both scales, and richness on farms embedded in complex relative to simple landscapes. Overall, both organic farming and in-field plant diversification exerted the strongest effects on pollinators and predators, suggesting these management schemes can facilitate ecosystem service providers without augmenting herbivore (pest) populations. Our results suggest that organic farming and plant diversification promote diverse arthropod meta-communities that may provide temporal and spatial stability of ecosystem service provisioning. Conserving diverse plant and arthropod communities in farming systems therefore requires sustainable practices that operate both within fields and across landscapes. |

1   A global synthesis of the effects of diversified farming systems on arthropod diversity within
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4   **Running head:** Effects of diversified farming on arthropods
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136   **Author contributions:**
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138   WES, NW, and RW; EML, DC, and CMK collected, prepared, and analyzed data and wrote
139   the manuscript; all authors except CMK and EML contributed empirical field data; all
140   authors revised the manuscript.

141  **ABSTRACT**
142
143  Agricultural intensification is a leading cause of global biodiversity loss, which can reduce

144  the provisioning of ecosystem services in managed ecosystems. Organic farming and plant

145  diversification are farm management schemes that may mitigate potential ecological harm by

146  increasing species richness and boosting related ecosystem services to agroecosystems. What

147  remains unclear is the extent to which farm management schemes affect biodiversity

148  components other than species richness, and whether impacts differ across spatial scales and

149  landscape contexts. Using a global meta-dataset, we quantified the effects of organic farming

150  and plant diversification on abundance, local diversity (communities within fields), and

151  regional diversity (communities across fields) of arthropod pollinators, predators, herbivores,

152  and detritivores. Both organic farming and higher in-field plant diversity enhanced arthropod

153  abundance, particularly for rare taxa. This resulted in increased richness but decreased

154  evenness. While these responses were stronger at local relative to regional scales, richness

155  and abundance increased at both scales, and richness on farms embedded in complex relative

156  to simple landscapes. Overall, both organic farming and in-field plant diversification exerted

157  the strongest effects on pollinators and predators, suggesting these management schemes can

158  facilitate ecosystem service providers without augmenting herbivore (pest) populations. Our

159  results suggest that organic farming and plant diversification promote diverse arthropod

160  meta-communities that may provide temporal and spatial stability of ecosystem service

161  provisioning. Conserving diverse plant and arthropod communities in farming systems

162  therefore requires sustainable practices that operate both within fields and across landscapes.

163     **INTRODUCTION**

164         Simplification of agricultural landscapes, and increased use of fertilizers and

165     pesticides, threaten arthropod communities worldwide (Matson *et al.*, 1997; Tscharntke *et al.*,

166     2005; Potts *et al.*, 2016). This could impair agricultural sustainability because declines in

167     arthropod abundance and diversity are often associated with reduced provisioning of

168     ecosystem services including pollination, pest control, and nutrient cycling (Kremen & Miles,

169     2012; Oliver *et al.*, 2015). Two strategies purported to mitigate this ecological harm are

170     organic farming and in-field plant diversification (Table S1). We refer to these strategies as

171     farm management schemes, both of which include a host of practices that promote biological

172     diversification (Kremen & Miles, 2012; Puech *et al.*, 2014). We refer to organic farming,

173     conventional farming, high in-field plant diversification, and low in-field plant diversification

174     as separate field types. Mounting evidence indicates that arthropod communities are more

175     diverse and abundant in fields lacking synthetic fertilizers and pesticides, and in those with

176     greater plant diversity (e.g., intercropped or having non-crop vegetation like hedgerows or

177     floral strips) (Letourneau *et al.*, 2011; Crowder *et al.*, 2012; Kennedy *et al.*, 2013; Garibaldi

178     *et al.*, 2014; Batáry *et al.*, 2015; Fahrig *et al.*, 2015).

179         The benefits of diversified farming practices may manifest at different scales, such as

180     within individual fields (local diversity) or across multiple fields in a landscape (regional

181     diversity) (Table S1). One observational study of 205 farms across Europe and Africa, for

182     example, found that although organic farming provided strong benefits for local richness of

183     plants and pollinators, these benefits faded at regional scales (Schneider *et al.*, 2014). This

184     suggests that while farmers may promote local diversity on their field(s) by using organic

185     practices, their efforts may not enhance biodiversity across multiple fields. Conversely, the

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186    addition of hedgerows to crop fields has been shown to increase community heterogeneity

187    and species turnover (measures of local diversity), which are important components of

188    regional diversity (Ponisio *et al.*, 2016). The effects of farm management for particularly

189    mobile arthropods, such as pollinators, may also transcend individual fields if the improved

190    quality of habitats on one field boosts abundance, with organisms spilling over to nearby

191    fields (Tscharntke *et al.*, 2012; Kennedy *et al.*, 2013). While increases in local diversity have

192    been shown to provide the strongest benefits to individual ecosystem services (i.e.,

193    pollination and biological control), regional diversity can support the simultaneous provision

194    of multiple ecosystem services over space and time (Pasari *et al.*, 2013). Thus, to mitigate the

195    effects of biodiversity loss across agroecosystems, farm management schemes should ideally

196    benefit both local and regional diversity.

197           Research on the impacts of organic farming and in-field plant diversity has primarily

198    focused on beneficial functional groups such as natural enemies and pollinators (Crowder *et*

199    *al.*, 2010; Kennedy *et al.*, 2013) across intensively sampled regions of Europe and North

200    America (Shackelford *et al.*, 2013; De Palma *et al.*, 2016). Moreover, almost all studies rely

201    on richness (the number of taxa; Table S1) as a proxy for biodiversity but ignore metrics such

202    as evenness (the relative abundances among species; Table S1) (e.g., Bengtsson *et al.*, 2005;

203    Tuck *et al.*, 2014). Yet, richness poorly reflects overall community diversity (Duncan *et al.*,

204    2015; Loiseau & Gaertner, 2015), and its measurement is strongly confounded by abundance

205    (Chao & Jost, 2012). Variation in richness has also been shown to have minimal impacts on

206    ecosystem functioning when richness increases are driven primarily by rare species that

207    contribute little to ecosystem services (Kleijn *et al.*, 2015; Winfree *et al.*, 2015). While

208    common species may provide the majority of ecosystem services on some farms (Schwartz *et*

209    *al.*, 2000; Kleijn *et al.*, 2015), rare species can provide redundancy (Kleijn *et al.*, 2015) or

210    support provisioning of multiple ecosystem services (Soliveres *et al.*, 2016). Assessing

211    evenness can help determine whether richness increases are driven by rare or common

212    species. Richness, evenness, and abundance can also independently or interactively affect

213    ecosystem function (Wilsey & Stirling, 2006; Wittebolle *et al.*, 2009; Crowder *et al.*, 2010;

214    Northfield *et al.*, 2010; Winfree *et al.*, 2015). Thus, teasing apart the effects of farm

215    management schemes on abundance and each diversity metric is critical. While existing

216    studies find that organic farming and in-field plant diversification tend to boost abundance

217    and richness of certain taxa, whether these effects are consistent for other biodiversity

218    components such as evenness, for functional groups other than pollinators and natural

219    enemies, and for less-well studied regions of the world (e.g., the tropics and Mediterranean)

220    remains unclear.

221        Here, we present a comprehensive synthesis of studies that explore how organic

222    farming and in-field plant diversification influence arthropod communities across global

223    agroecosystems. We determine whether community responses to these management schemes

224    vary based on different metrics (abundance, local richness and evenness, regional richness

225    and evenness) and arthropod functional groups (detritivores, herbivores, pollinators, and

226    predators). We investigate if these responses depend on landscape complexity (i.e., the

227    proportion of natural and semi-natural habitat surrounding the farm; Fig. S1, Table S1),

228    because landscape heterogeneity has been shown to influence the effectiveness of farm

229    management schemes (Batáry *et al.*, 2011; Kleijn *et al.*, 2011; Kennedy *et al.*, 2013; Tuck *et*

230    *al.*, 2014). We also explore whether farm management schemes have similar impacts on

231    relatively rare compared to common taxa. Our results demonstrate whether local and regional

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232   diversity and abundance of different functional groups are similarly affected by on-farm

233   management and landscape complexity, and the extent of covariance between biodiversity

234   within and across fields in a landscape. Broadly, our findings further reveal the role of farm

235   management in mitigating biodiversity loss and maintaining healthy arthropod communities

236   in agroecosystems under global change.

237

238   **MATERIALS AND METHODS**

239   **Literature survey**

240          We compiled data from studies on arthropod diversity in agroecosystems that

241   compared one or both of the farm management schemes of interest: (1) organic vs.

242   conventional farming and (2) high vs. low in-field plant diversity. We defined organic

243   agriculture as fields that were organically certified or met local certification guidelines (Table

244   S1). These guidelines involve, at minimum maintaining production systems free of synthetic

245   pesticides and fertilizers. We defined conventional agriculture as fields or farms that used

246   recommended rates of synthetic, or a mix of synthetic and organic, pesticides and fertilizers.

247   Other types of farming systems, such as integrated, which fit neither category where excluded

248   from the analysis. Fields were defined as having high in-field plant diversity if they had

249   diverse crop vegetation or managed field margins to include non-crop vegetation (e.g.,

250   hedgerows, border plantings, flower strips) (Table S1). We also classified small (< 4 ha)

251   fields as diverse because they yield small-scale crop diversity (across several fields) even if

252   the target field is a monoculture (Pasher *et al.*, 2013). Fields were defined as having low in-

253   field plant diversity if they had none of these features. Studies that compared these schemes

254   were identified by (1) searching the reference lists of recent meta-analyses (Batáry *et al.*,

255   2011; Chaplin-Kramer *et al.*, 2011; Crowder *et al.*, 2012; Garibaldi *et al.*, 2013; Kennedy *et*

256   *al.*, 2013; Scheper *et al.*, 2013; Shackelford *et al.*, 2013), (2) searching ISI Web of

257   Knowledge (April and May 2013) using the terms "evenness or richness" and "organic and

258   conventional" or "local diversity", and (3) directly contacting researchers who study

259   arthropods in agricultural systems.

260          We identified 235 relevant studies that we examined for inclusion based on five

261   criteria: (1) sampling was performed in the same crop or crop type (e.g., cereals) for organic

262   and conventional fields, or fields with high and low in-field plant diversity; (2) sampling was

263   conducted at the scale of individual crop fields rather than using plots on experiment stations;

264   (3) the study included at least two fields of each type; (4) all organisms collected were

265   identified to a particular taxonomic level (i.e., order, family, genus, species, or

266   morphospecies), such that no taxa were lumped into groups such as "other"; and (5) at least

267   three unique taxa were collected. We use "taxon" to refer to a single biological type (e.g.,

268   species, morphospecies, genus, family), determined as the finest taxonomic resolution to

269   which each organism was identified in a particular study (see examples in Table S1). A total

270   of 60 studies met our criteria, representing 43 crops, 21 countries, and 5 regions (Asia,

271   Europe, North and Central America, South America, Oceania) (Fig. S2, Table S2). For

272   studies that investigated both management scheme comparisons, we included the data in both

273   analyses only when the field types were independently assigned (Table S3); otherwise we

274   selected the scheme that the authors indicated the study was designed to address (Table S2).

275   Across these 60 studies, our meta-analysis included 110 unique data points: 81 comparing

276   organic and conventional fields and 29 comparing fields with high vs. low in-field plant

277   diversity (Fig. S2, Tables S2, S4, archived data). Among organic vs. conventional studies, the

278    number with high in-field plant diversity, low in-field plant diversity, and both levels of plant

279    diversity was independent of organic vs. conventional management ($\chi^2_2 = 0.47$, $p = 0.79$).

280

281    **Calculation of effect sizes**

282    Unlike traditional meta-analyses that extract summary statistics from studies, we

283    gathered and manipulated raw data, which enabled us to calculate evenness and classify taxa

284    into functional groups. For each study, we compiled data on the abundance of all taxa in each

285    field. For studies conducted across multiple years or crop types, separate values were

286    compiled for each year and crop. To avoid pseudoreplication, for multi-year studies we

287    selected a single year to analyze based on maximizing the number of (1) sites that met the

288    evenness criterion (at least three taxa), (2) fields, or (3) individuals (in decreasing priority

289    order; Garibaldi *et al.*, 2013). Each collected taxon was classified into one of four functional

290    groups: detritivore, herbivore, pollinator, or predator (see Supporting Methods for details).

291    These taxon-level data were used to calculate effect sizes for abundance, local diversity, and

292    regional diversity in paired organic vs. conventional or high vs. low in-field plant diversity

293    systems. For local and regional calculations, we defined diversity as both richness and

294    evenness, and treated each functional group separately (Fig. S1).

295    Local diversity reflects the average diversity within each field, and was calculated

296    using individual crop fields as the sampling unit (Fig. S1, Table S1). In studies with sub-

297    samples at a scale smaller than a field (i.e., plots within fields), values across these sub-

298    samples were averaged before calculating local diversity. Abundance was the number of

299    arthropods, and richness the number of unique taxa, in a field. Evenness was calculated using

300    the metric $E_{var}$, which ranges from 0 (one taxon dominant) to 1 (uniform abundance for all

301  taxa). This metric was chosen for its desirable statistical properties, particularly independence

302  from richness, and its use in similar previous meta-analyses (Crowder *et al.*, 2012). After

303  calculating abundance, richness, and evenness for each field, we averaged values across all

304  fields of a particular type in a study to obtain the values for effect size calculations.

305       Regional diversity values were calculated based on individuals pooled across all fields

306  in a study (Fig. S1, Table S1). Thus, regional richness and evenness are measures of diversity

307  of meta-communities across fields in a landscape, while local diversity measures

308  communities in a single field (Wang & Loreau, 2014). We note that regional diversity is not a

309  direct indication of spatial scale, as the geographical extent of sampling varied among

310  studies. Some studies were not designed to assess regional diversity specifically, and sampled

311  unequal numbers of fields of each type. To correct for this sampling bias, we used sample-

312  based rarefaction with 1,000 random samples taken from the set of fields in a given study to

313  determine pooled species assemblages (Gotelli & Colwell, 2011). For example, if a study had

314  10 conventional and 6 organic fields, regional diversity values for the conventional

315  management schemes would be based on the average pooled community taken from 1,000

316  random draws of 6 field sites. Regional abundance is simply local abundance multiplied by

317  the number of sites, thus we reported only one abundance value per study.

318       To compare effects of farm management schemes on diversity and abundance, we

319  used the log-response ratio as an effect size metric (Hedges *et al.*, 1999). We used this metric,

320  rather than a weighted effect size, for three reasons. First, weighted effect sizes could not be

321  calculated for regional diversity because these calculations were based on a single value

322  (without replication) from each study, such that there was no estimate of variability. Second,

323  our studies classified arthropods at varying levels of taxonomic resolution. Studies classified

324    at the family level had less variability than studies classified at the species level, so using a

325    weighted metric would give studies conducted at a coarser taxonomic resolution greater

326    weight. Finally, preliminary analysis showed weighted and unweighted analyses of local

327    diversity and abundance were qualitatively similar (Table S5). In the Results, we back-

328    transformed log response-ratio effect sizes to percentages.

329          We assessed funnel plot asymmetry to test for publication bias. Because we used an

330    unweighted effect size metric, we plotted effect sizes against sample sizes (i.e., number of

331    fields; Figs. S3, S4) (Sterne & Egger, 2001), and visually assessed asymmetry since formal

332    statistical tests require effect size variances (Jin *et al.*, 2015) and measures of regional

333    diversity had no variance component. Visual assessment looked for, and did not find, areas of

334    missing non-significant results, a directional bias to effects, or a strong relationship between

335    effect and sample sizes. We did not detect any sign of publication bias; funnel plots were

336    sufficiently symmetrical. Finally, we ensured the sampling method (active versus passive

337    sampling techniques) did not influence results (see Supporting Information, Table S6). We

338    calculated abundance and diversity values with R v. 3.1.1 (R Core Team, 2014), using

339    packages BiodiversityR (Kindt & Coe, 2005), doBy (Højsgaard & Halekoh, 2013), and

340    reshape (Wickham, 2007).

341

342    **Study variables**

343          We gathered data on three categorical variables and assessed whether they mediated

344    arthropod responses to farm management schemes: (1) landscape complexity (simple,

345    complex), (2) biome (boreal, Mediterranean, temperate, tropical), and (3) crop cultivation

346    period (annual, perennial). Landscape complexity (see Fig. S1, Table S1) was determined

347    from land cover data on the percentage of natural and semi-natural habitat within 1 km of

348    sampled fields. Natural and semi-natural habitat was defined as areas dominated by forest,

349    grassland, shrubland, wetlands, ruderal vegetation, or non-agricultural plantings (i.e.,

350    previously-cultivated areas where vegetation is regenerating, hedgerows, field margins, and

351    vegetation along roadways or ditches). For each study, we calculated the mean percentage of

352    natural habitats across fields using locally-relevant land cover databases. Landscapes were

353    classified as simple if they averaged ≤ 20% natural habitat, and complex if they averaged >

354    20% natural habitat, following Tscharntke et al. (2005) and common practice (e.g., Batáry *et*

355    *al.*, 2011; Scheper *et al.*, 2013) (see Supporting Methods for additional details). Biome was

356    based on the geographic location of the study. Crop cultivation periods were derived from

357    several sources (FAO AGPC, 2000; Garibaldi *et al.*, 2013). Table S4 shows the distribution

358    of data points across each of these descriptive variables.

359

360    **Data analyses**

361         Table S7 summarizes specific questions we addressed and the approach we used to

362    test each one. We first used one-sample *t*-tests (Crowder & Reganold, 2015) to determine if

363    the mean effect sizes for abundance, local richness and evenness, and regional richness and

364    evenness differed significantly from 0. For each management scheme comparison (organic

365    vs. conventional or high vs. low in-field plant diversity), these analyses were conducted for

366    the overall arthropod community and for each functional group separately. We also explored

367    correlations between local and regional richness, and between local and regional evenness, to

368    determine if these metrics responded similarly to each of the management schemes. We used

369    $\alpha = 0.10$, to describe effect sizes that appeared ecologically important but did not meet the

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370    somewhat arbitrary α = 0.05. This accords with a recent policy statement by the American

371    Statistical Association (Wasserstein & Lazar, 2016), which notes that reliance on arbitrary

372    alpha values can lead to erroneous conclusions.

373        In subsequent analyses, we used meta-regression to examine whether effect sizes

374    were influenced by functional group and other study characteristics. We excluded studies

375    lacking landscape complexity data (see archived data) from meta-regressions. For each

376    management scheme and response, we ran a linear mixed model (lme4 package; Bates *et al.*,

377    2014) that included eight fixed effect variables: (1) functional group (detritivore, herbivore,

378    predator, pollinator), (2) diversity scale (local, regional), (3) landscape complexity (simple,

379    complex), (4) biome (boreal, Mediterranean, temperate, tropical), (5) crop cultivation period

380    (annual, perennial), (6) functional group×diversity scale interaction; (7) functional

381    group×landscape complexity interaction; and (8) diversity scale×landscape complexity

382    interaction. These models included study ID as a random effect. We used information-

383    theoretic model selection to determine the set of best-fit models for each response variable

384    (MuMIn package; Barton, 2014), which contained models with AICc values within 2 of the

385    smallest value (Burnham & Anderson, 1998). We examined significance of the fixed effects

386    in each model in the best-fit set (α = 0.10) with likelihood ratio tests, and used post-hoc

387    planned contrasts (with *p*-values adjusted to control the overall Type I error rate using

388    Holm's sequential Bonferroni procedure; see Supporting Methods) (phia package; Rosario-

389    Martinez, 2013) to test for (1) differences in effect size among functional groups and biomes,

390    (2) differences in effect size between the local and regional scales within each functional

391    group, and (3) landscape complexity differences between each pair of functional groups.

392    We also tested whether abundance and richness effect sizes differed for rare and

393    common taxa. Following Kleijn et al. (2015), within each study we classified taxa as

394    common if their relative abundance was at least 5% of the total community; other species

395    were categorized as rare. We then calculated local abundance and richness as well as regional

396    abundance and richness separately for rare and common taxa. We used one-sample *t*-tests to

397    determine if mean effect sizes differed significantly from zero, and paired *t*-tests to determine

398    whether mean effect sizes differed between rare and common taxa.

399

400    **RESULTS**

401    **Effects of management schemes on overall arthropod communities**

402    Organic farming increased arthropod abundance (45% change), local richness (19%),

403    and regional richness (11%) (Fig. 1a, Table S8). These positive effects were stronger for local

404    compared to regional richness (Fig. 1a, Tables S9, S10). Arthropod communities on organic

405    farms had significantly but only moderately lower local evenness (-6%) and regional

406    evenness (-8%) than on conventional farms (Fig. 1a, Table S8). Fields with high in-field plant

407    diversity increased local richness (23%) and regional richness (19%), with similar magnitude

408    (Fig. 1b, Tables S8, S11, S12). In-field plant diversity did not significantly affect abundance

409    (27%), local evenness (-6%) or regional evenness (-13%) (Fig. 1b, Table S8). Overall, there

410    were strong positive correlations between local and regional richness ($r = 0.87$), and between

411    local and regional evenness ($r = 0.57$; Fig. S5).

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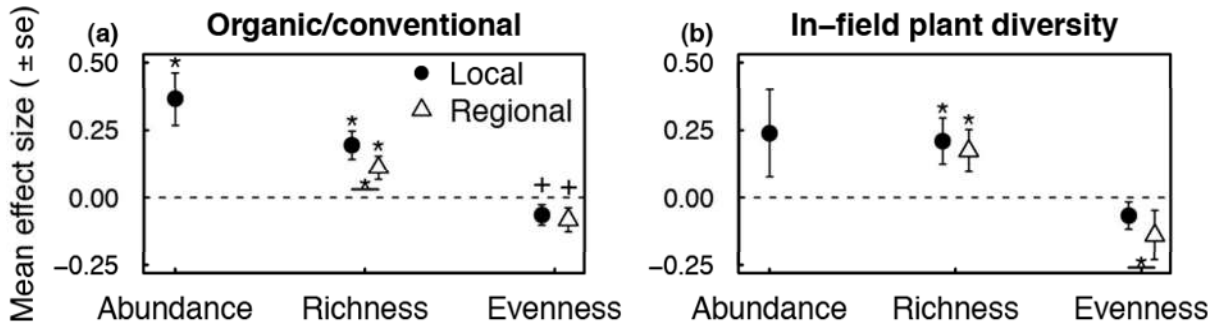
**Figure 1.** Effects of farm management schemes on arthropod abundance, local diversity, and regional diversity. Values shown are for the entire arthropod community, and represent the mean log-response ratio ($\pm$ SE) of (a) adopting organic farming and (b) promoting in-field plant diversity on abundance, richness, and evenness. A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$) above a mean denotes a significant difference from zero (determined via one-sample $t$-tests; statistical details in Table S8), while one below a pair of means indicates a significant difference between local and regional diversity (determined via linear mixed models; Tables S9-S12).

Organic farming increased abundance and richness of both rare and common arthropods at the local and regional scales (Fig. S6a,c, Table S13). At the local scale, organic farming increased arthropod richness by promoting rare taxa (27% increase) more strongly than common taxa (14% increase) (Fig. S6c, Table S14). In-field plant diversification also had differential effects on rare and common taxa, increasing richness of both at the local scale, but only of rare taxa at the regional scale (Fig. S6d, Table S13). Fields with higher in-field plant diversity increased abundance of common arthropods, but not of rare arthropods (Fig. S6b, Table S13).

**Effects of management schemes on arthropod functional groups**

Organic farming substantially increased the abundance (90%), local richness (55%), and regional richness (32%) of pollinator communities, but did not impact pollinator evenness (Fig. 2a, Table S15). For predator communities, organic farming increased abundance (38%) and local richness (14%), lowered local (-9%) and regional (-14%)

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437   evenness (Fig. 2c, Table S16), but did not affect regional richness (Fig. 2c, Table S16).

438   Organic farming also did not impact abundance, local or regional richness, or local or

439   regional evenness for herbivore (Fig. 2e, Table S17) or detritivore (Fig. 2g, Table S18)

440   communities. For all biodiversity components and functional groups, effect sizes in response

441   to organic farming did not differ between the local and regional scales (Fig. 2a,c,e,f, Tables

442   S9, S10). The diversity scale×landscape complexity interaction was never retained in a best-

443   fit model (Tables S9, S11).

444         High in-field plant diversity promoted the abundance (45%), local richness (44%),

445   and regional richness (29%) of pollinator communities, but decreased local pollinator

446   evenness (-11%) (Fig. 2b, Table S15). In-field plant diversity did not affect regional

447   pollinator evenness (Fig. 2b, Table S15). In addition, in-field plant diversity did not alter

448   abundance, local or regional richness, or local or regional evenness for predator (Fig. 2d,

449   Table S16) or herbivore (Fig. 2f, Table S17) communities. In-field plant diversity increased

450   the regional richness (69%) of detritivores and lowered regional detritivore evenness (-65%),

451   but did not impact detritivore abundance, local richness, or local evenness (Fig. 2h, Table

452   S18). The low sample size for detritivores, however, limits our ability to make inferences

453   about this group.

454

455   **Effects of landscape complexity, biome, and crop cultivation period on arthropod**

456   **communities**

457         Landscape complexity did not mediate the influences of organic farming or in-field

458   plant diversity on arthropod abundance or evenness (Fig. 3, Tables S9-S12). However, both

459   management schemes had stronger positive effects on local and regional arthropod richness

460    in complex relative to simple landscapes: organic farming 26% vs. 9%, in-field plant

461    diversification 29% vs. 11%, respectively (Fig. 3c,d, Tables S9-S12). The effects of

462    landscape complexity were similar in both direction and magnitude for local and regional

463    diversity (Fig. 3c-e, Tables S9-S12). Organic farming promoted herbivore richness to a

464    greater extent in simple than complex landscapes (Table S10), but other effects of landscape

465    complexity on abundance and diversity were similar across functional groups (Tables S9-

466    S12).

467         Stronger richness gains in complex than simple landscapes were driven

468    predominantly by rare taxa (Fig. 4). In complex landscapes, both organic farming and in-field

469    plant diversification had stronger positive effects on local richness of rare (organic 44%,

470    plant diversification 68%) than of common (organic 21%, plant diversification 18%)

471    arthropod taxa (Fig. 4c,d, Table S19). Organic farming within complex landscapes also

472    increased local abundance and regional richness of rare taxa (78% and 17%, respectively) to

473    a greater extent than common taxa (33% and 4%, respectively) (Fig. 4a, Table S19). Neither

474    management scheme differentially affected abundance or richness of rare and common taxa

475    in simple landscapes (Fig. 4, Table S19).

476         Biome mediated the impacts of in-field plant diversity on arthropod richness (pooled

477    across local and regional scales) (Tables S11, S12). Post-hoc tests failed to indicate

478    significant differences among biomes when considering all studies; but when the single

479    boreal study was removed from the analysis, high in-field plant diversity more strongly

480    promoted richness in Mediterranean (53%) than in temperate studies (-2%) (Table S12).

481    Biome did not mediate the effects of organic farming or in-field plant diversification on

482    arthropod abundance or evenness (Tables S9-S12). Organic farming increased arthropod

483    abundance to a greater extent in annual (70%) than in perennial (1%) crops (Tables S9, S10).

484    The effects of in-field plant diversification on abundance and diversity were consistent across

485    crop cultivation periods (Tables S11, S12).

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489  **Figure 2.** Effects of farm management schemes on abundance, local diversity, and regional
490  diversity of arthropod functional groups. Mean log-response ratios ($\pm$ SE) of (left column)
491  adopting organic farming and (right column) promoting in-field plant diversity for (a-b)
492  pollinators, (c-d) predators, (e-f) herbivores, and (g-h) detritivores. A "*" ($p < 0.05$) or "+"
493  ($0.05 \leq p < 0.1$) above a mean denotes a significant difference from zero (determined via one-
494  sample $t$-tests; Tables S15-S18). Meta-regressions indicated that differences between local
495  and regional values did not vary with functional group (Tables S9-S12).
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499   **Figure 3.** Effects of landscape complexity on the entire arthropod community in organic vs.
500   conventional farms (left column) and fields with high vs. low in-field plant diversity (right
501   column). Each graph shows the mean log-response ratio (± SE) for studies in simple (≤ 20%
502   natural habitat) or complex (>20% natural habitat) landscapes for (a,b) abundance, (c,d)
503   richness, and (e,f) evenness. A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$) below a set of means
504   indicates a significant difference between means at the habitat complexity levels (Tables S9-
505   S12).
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509 **Figure 4.** Effects of farm management schemes on abundance (a, b) and richness (c, d) of
510 common vs. rare taxa in simple and complex landscapes. Mean log-response ratios (±SE) of
511 (left column) adopting organic farming and (right column) promoting in-field plant diversity.
512 A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$) below a pair of means indicates a significant
513 difference between rare and common taxa within a landscape complexity category
514 (determined via paired $t$-tests; Table S19).
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516
517 **DISCUSSION**

518          Our global meta-analysis showed that both organic farming and in-field plant

519 diversification strongly increased arthropod abundance and richness, but had weaker effects

520 on evenness. The minimal evenness decreases on diversified farms reflected the presence of

521 more rare taxa. Emerging evidence suggests that rare taxa contribute to individual ecosystem

522 services less than common taxa (Schwartz *et al.*, 2000; Kleijn *et al.*, 2015), although they

523   may be important for maintenance of multiple ecosystem services across time and space

524   (Isbell *et al.*, 2011; Soliveres *et al.*, 2016). Thus, while organic farming and plant

525   diversification promote arthropod biodiversity conservation goals, their impacts on

526   ecosystem services may be nuanced. The positive effects of both organic farming and in-field

527   plant diversification were greatest for two groups of beneficial arthropods: pollinators and

528   predators. Thus, both schemes may increase agroecosystem sustainability by promoting key

529   ecosystem service providers without boosting pest (herbivore) densities.

530        Previous meta-analyses have investigated how organic farming and, to a lesser extent,

531   in-field plant diversification, affect arthropod abundance and richness (e.g., Bengtsson *et al.*,

532   2005; Batáry *et al.*, 2011; Chaplin-Kramer *et al.*, 2011; Scheper *et al.*, 2013; Shackelford *et

533   al.*, 2013; Tuck *et al.*, 2014). Our study extends upon this work by (1) combining data on

534   multiple arthropod functional groups (but see Shackelford *et al.*, 2013), and (2) examining

535   the type and scale of diversity across a variety of crop types. As such, we offer a more

536   comprehensive understanding of when and how farm management schemes alter arthropod

537   biodiversity. Our findings caution that the frequent use of richness as the sole proxy for

538   biodiversity fails to reflect the full impacts of farming practices on biologic communities.

539   While multiple studies have shown that organic farming boosts richness (e.g., Bengtsson *et

540   al.*, 2005; Tuck *et al.*, 2014), we found that evenness decreased, an outcome that was due

541   mainly to promotion of rare species. Species richness might be increased by conservation

542   practices that target specific taxa, but the promotion of evenness requires practices that can

543   simultaneously balance the abundances of many taxa (Crowder *et al.*, 2010, 2012). Finally,

544   our results highlight the necessity of targeting farm management within the context of local

545   conditions (Cunningham *et al.*, 2013; Saunders *et al.*, 2016). For example, our results suggest

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546    that farmers in Mediterranean biomes might see greater arthropod richness gains by

547    increasing in-field plant diversity than by farming organically, while farmers growing annual

548    crops may be more likely to boost arthropod abundance with organic farming.

549        Disentangling relationships between biodiversity components at local and regional

550    scales can inform patterns of community assembly and mechanisms that shape community

551    structure (Gering & Crist, 2002; Wang & Loreau, 2014). We found that regional diversity

552    positively correlated with local diversity under both management schemes. Further, organic

553    farming increased richness at both scales, although local effects were stronger than regional

554    ones. One possible explanation is that diversified farming practices increase the heterogeneity

555    of local communities (e.g., Ponisio *et al.*, 2016), which could lead to greater regional

556    diversity. Another possibility is that diversified fields serve as source habitats within a matrix

557    of crop and non-crop habitats across farming landscapes (M'Gonigle *et al.*, 2015). Further,

558    the benefits of diversification practices on local communities in fields can be strongly

559    mediated by regional species pools across farming landscapes (Gering & Crist, 2002).

560        Our results, in combination with another recent meta-analysis (Schneider *et al.*, 2014),

561    suggest that mobility of organisms can determine whether the benefits of farm diversification

562    accrue at both local and regional scales. While we show that organic farming can boost

563    arthropod diversity at local and regional scales, Schneider et al. (2014) found that organic

564    farming increased plant, earthworm, and spider richness at field but not regional scales.

565    These groups of organisms tend to have limited dispersal capacity, particularly plants and

566    earthworms. Thus, their local communities may be structured more by competition than long-

567    distance dispersal (Gering & Crist, 2002), which would limit the similarity between

568    communities within and across fields. At the same time, Schneider et al. (2014) found that

569 organic farming boosted the richness of bees, a more mobile group of organisms, by

570 approximately 25% at the local scale and 15% at the regional scale. We likewise found that

571 diversified farming increased abundance, and local and regional richness, of mobile

572 pollinators, but had less impact on detritivores that tend to have lower mobility (Sattler *et al.*,

573 2010).

574        Overall, our results are consistent with mounting evidence that farm management and

575 landscape complexity interactively affect arthropod biodiversity (e.g., Rusch *et al.*, 2010;

576 Batáry *et al.*, 2011; Kennedy *et al.*, 2013; Tuck *et al.*, 2014), although results across studies

577 have found sometimes conflicting patterns (Kleijn *et al.*, 2011; Tscharntke *et al.*, 2012; Tuck

578 *et al.*, 2014). For example, agri-environment schemes that promote low input, low

579 disturbance, and diverse farms are sometimes most effective in fostering biodiversity in

580 structurally simple landscapes (Batáry *et al.*, 2011; Scheper *et al.*, 2013). This presumably

581 occurs because simple landscapes fail to satisfy the resource needs of many species, such that

582 these species may disperse into diverse farms to seek resources (Tscharntke *et al.*, 2005;

583 Kremen & Miles, 2012). In contrast, we found that impacts of organic farming and plant

584 diversification on arthropod richness were heightened for fields embedded in complex

585 landscapes. This could occur if complex landscapes support more diverse species pools that

586 can respond positively to farm management (Duelli & Obrist, 2003; Hillebrand *et al.*, 2008;

587 Kennedy *et al.*, 2013). Consistent with this hypothesis, we showed that organic farming in

588 complex landscapes preferentially increased richness of rare taxa locally (i.e., in fields) and

589 regionally (i.e., across landscapes). Importantly, the interactive effects of landscape

590 complexity and on-farm management may differ across arthropod functional groups with

591 varying capacity to move across landscapes (Tscharntke *et al.*, 2005; Chaplin-Kramer *et al.*,

592    2011). However, the only interaction between landscape complexity and management

593    schemes we found was for richness of herbivores, a group with considerable variation in

594    mobility among taxa (Sattler *et al.*, 2010).

595         Ideally, increases in abundance and diversity of arthropods on farms would enhance

596    the provisioning of ecosystem services (Kremen & Miles, 2012). However, empirical studies

597    have provided mixed evidence. In-field plant diversification and increased landscape

598    complexity have been found to promote predator abundance and diversity with no change in

599    pest control levels (Chaplin-Kramer *et al.*, 2011; Rusch *et al.*, 2016) or reduced crop damage

600    (Letourneau *et al.*, 2011). The relationship between biodiversity and ecosystem services on

601    farms is thus likely strongly mediated by species' abundances and functional roles. For

602    example, Northfield *et al.* (2010) found that greater predator richness increased pest control,

603    but only with high predator densities where complementarity among predator species was

604    fully realized. Increases in pollinator richness can have minimal impacts on ecosystem

605    services when richness gains are associated with rare species that contribute little to

606    pollination (Kleijn *et al.*, 2015; Winfree *et al.*, 2015). Increasing wild pollinator richness on

607    large farms (> 14 ha) only increases fruit set when wild pollinator density is also high

608    (Garibaldi *et al.*, 2016). Higher predator species evenness on organic farms has also been

609    shown to translate to increased pest control, with the potential to reduce yield gaps compared

610    with conventional agriculture (Crowder *et al.*, 2010). However, models suggest that

611    decreased evenness could also lead to greater ecosystem services when abundance of

612    common species that are effective ecosystem services providers increases at the expense of

613    rare species that are functionally less important (Crowder & Jabbour, 2014), a result seen

614    with pollinators in agricultural systems (Kleijn *et al.*, 2015; Winfree *et al.*, 2015). The

615     combination of context-specific responses to farm management schemes shown by this study

616     and biodiversity-ecosystem functioning relationships that depend on species' abundances and

617     functional traits suggest that the effects of diversified farming on ecosystem services are

618     likely to depend on biome, landscape, and crop characteristics.

619        By promoting biodiversity and abundance of arthropods, diversified agriculture could

620     provide a multitude of other benefits (Oliver *et al.*, 2015). Biodiversity can help maintain

621     stability of ecosystem processes through mechanisms such as response diversity and

622     functional redundancy (Cardinale *et al.*, 2012; Mori *et al.*, 2013). Arthropod richness gains in

623     response to organic farming and plant diversification, such as those documented here, could

624     guard against the loss of ecological function by supporting multiple species that occupy

625     similar functional niches (functional redundancy) or that are functionally similar but respond

626     differentially to environmental change (response diversity; Elmqvist *et al.*, 2003). The

627     abundance and richness increases we detected for pollinators and predators but not for

628     herbivores suggest that the two former groups may benefit more from these stabilizing

629     processes. Resilient systems must also exhibit multiple ecosystem functions

630     (multifunctionality) as environmental conditions and arthropod populations fluctuate.

631     Increases in rare taxa, as detected in this study, may be critical for multifunctionality (Isbell

632     *et al.*, 2011; Soliveres *et al.*, 2016) and even for single ecosystem functions (Zavaleta &

633     Hulvey, 2004; Mouillot *et al.*, 2013). Thus, regional-scale refuges for rare species may ensure

634     resilient agricultural systems.

635        Overall, our results suggest that organic farming and in-field plant diversification both

636     promote biodiversity on farms. Moreover, these two schemes might have interactive effects

637     on farm productivity. Practices such as multi-cropping (plant diversification) and longer,

638     more diverse, crop rotations can reduce the yield gaps between organic and conventional

639     agriculture (Ponisio *et al.*, 2015), and increase the profitability of organic relative to

640     conventional systems (Crowder & Reganold, 2015). Diversified small farms are increasingly

641     being replaced by large, simplified, and intensive monoculture production systems

642     (Tscharntke *et al.*, 2005; Bennett *et al.*, 2012). This is problematic because intensified

643     farming reduces the long-term sustainability of agroecosystems, thereby threatening global

644     food security (Ray *et al.*, 2012). One of the greatest challenges of the 21$^{st}$ century is meeting

645     the food, fiber, and energy needs of a growing human population while maintaining farm

646     sustainability and ecosystem functioning (Tilman *et al.*, 2011). Our study underscores that

647     adopting organic farming or in-field plant diversification practices might aid society in

648     attaining these goals.

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656     **DATA ACCESSIBILITY**

657          Data and scripts available at: [insert DOI for Zenodo repository]

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849

1   **SUPPORTING METHODS**
2
3   **Functional group classifications**
4       Data providers determined the functional group of each taxon. When functional groups
5   were non-defined or non-standard (e.g., saprophage), or when taxa filled multiple functional
6   roles (e.g., species that serve as both pollinators and herbivores), we assigned taxa to a single
7   functional role based on their most common description in the literature. To maximize data
8   inclusion, we also (1) combined predators and parasitoids, (2) classified all carabids as predators
9   since even the herbivorous species are thought to consume some animal material (e.g.,
10  Hengeveld, 1980; Jørgensen & Toft, 1997), and (3) classified a few pollinators as herbivores in
11  studies with few pollinator taxa but many herbivores.
12
13  **Sampling methods**
14      Studies used a broad range of sampling methods, which we categorized as active or
15  passive. Active sampling methods included beating, netting bees seen at plants, hand-collecting
16  individuals off plants, observational counting, washing plants, taking soil cores, sweep-netting,
17  and vacuum sampling. Passive sampling methods were blue vane traps, light traps, visually-
18  attractive or scented lures, malaise traps, minnow traps, pan traps, pitfall traps, and sticky cards.
19  However, we did not include sampling method in our meta-regressions because preliminary
20  analyses indicated that sampling method negligibly affected effect sizes (Table S5).
21
22  **Landscape complexity**
23      The "simple" landscape complexity category combined Tscharntke et al.'s (2005)
24  "cleared" and "simple" categories because we had only two "cleared" studies. We were unable to
25  categorize landscape complexity when we obtained data directly from published articles that
26  lacked GPS coordinates of sampling locations or information on natural habitat surrounding
27  fields (Study IDs drit01, febe01, hesl01, hokk01, and weib01). These five studies all compared
28  organic and conventional farms. In a couple of cases we based landscape complexity on
29  percentage of natural habitat within 500 m (bosq01), or the average of percentages at 500m and
30  1.5 km (leto01; percentages at the two distances strongly correlated, with $r = 0.8$).
31
32
33  **SUPPORTING REFERENCES**
34

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42      dichotomy: Does it make sense for natural enemies? *Agriculture, Ecosystems &*
43      *Environment*, **194**, 48–57.
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45      on agricultural intensification and biodiversity – ecosystem service management. *Ecology*
46      *Letters*, **8**, 857–874.

47    **Table S2 is in a separate file.**

48

49    **Table 1:** Definitions and descriptions of key terms.

50

| Term | Definition | Notes |
|---|---|---|
| Organic farming | Organically certified, or meeting local certification guidelines. While guidelines vary by country, they typically involve, at minimum, maintaining production systems free of synthetic pesticides and fertilizers. | Both organic and conventional farming include a broad range of management strategies and levels of intensity (e.g., pesticide application frequency, monoculture vs. polyculture) (Kremen & Miles, 2012; Puech *et al.*, 2014). |
| Conventional farming | Fields or farms that used recommended rates of synthetic, or synthetic and organic, pesticides and fertilizers. | |
| In-field plant diversification | This includes various schemes that increase small-scale plant diversity, including intercropping, managing field margins to include non-crop vegetation (e.g., hedgerows, border plantings, flower strips), and use of small (< 4 ha) fields. | |
| Taxon | A single biological type (e.g., species, morphospecies, genus, family), determined as the finest taxonomic resolution to which each organism was identified. | Examples: *Apis mellifera* (species), *Halictus* sp. 1 (morphospecies), *Lasioglossum* spp. (genus), Formicidae (family). We assigned each taxon to a functional group (detritivore, herbivore, pollinator, predator), but calculated abundance and diversity from taxon-level data. |
| Abundance | The number of total individuals, of all taxa together, sampled. | We calculated abundance, richness, and evenness separately for each field type (e.g., conventional farming), crop, year, and arthropod functional group within each study. |
| Richness | The number of taxa sampled. | |
| Evenness | How individuals are distributed across taxa in the sample. The evenness measure that we used, $E_{var}$, range from 0 (completely uneven, one taxon dominates) to 1 (completely even, with each taxon represented by an equal number of individuals. | |
| Region | A large spatial extent that contains multiple communities and habitats. We defined each study's region as | |

| | all of the fields sampled in the study. | |
|---|---|---|
| Rare taxon | A taxon with relative abundance less than 5% of all individuals sampled across the region. | We determined rarity separately for each management scheme comparison (organic vs. conventional, high vs. low in-field plant diversity), crop, year, and function within a study, but did not further separate by field type. |
| Local diversity | Diversity (here, richness and evenness) of a community within a field. | We estimated local abundance and diversity by first calculating abundance and diversity values within each field, then averaging these values across fields. For example, assume species A, B, C, D, and E were found in field 1; species A, E, and F in field 2; species B, C, D, and E in field 3; and species A, B, E, F, G, and H in field 4. Each field's richness would be 5, 3, 4, and 6, respectively. Local richness would be 4.5, the average of each field's richness value. |
| Regional diversity | Diversity (here, richness and evenness) of the meta-community that spans all fields in a region. | We estimated regional diversity by pooling individuals sampled in all fields within a landscape, then calculating diversity of taxa in this one regional sample. In the above example, the regional species pool would include species A through H and regional richness would be 8. |
| Landscape complexity | The proportion of natural and semi-natural habitat (areas dominated by forest, grassland, shrubland, wetlands, ruderal vegetation, or non-agricultural plantings including previously-cultivated areas where vegetation is regenerating, hedgerows, field margins, and vegetation along roadways or ditches) surrounding a farm. | We determined landscape complexity separately for each management scheme comparison, crop, and year within a study, by averaging proportions across fields. |

51

52 **Table S3.** Fisher exact tests for studies with variation in both management (organic vs.
53 conventional) and in-field plant diversity (high vs. low). These tests were used to determine
54 whether sites were assigned independently to management types across the two management
55 schemes. I-f=in-field plant diversity
56

| Study ID | Number of sites with: | | | | *p*-value | Management scheme(s) used |
|---|---|---|---|---|---|---|
| | Organic & high i-f | Organic & low i-f | Conventional & high i-f | Conventional & low i-f | | |
| bomm01 | 8 | 16 | 22 | 53 | 0.80 | Both |
| bosq01 | 7 | 10 | 10 | 10 | 0.74 | Both |
| clou01 | 15 | 6 | 10 | 11 | 0.21 | Both |
| danf01 | 2 | 0 | 3 | 5 | 0.44 | Both |
| eige01 | 3 | 0 | 0 | 3 | 0.10 | Organic/ conventional |
| ekro01 | 7 | 8 | 12 | 4 | 0.15 | Both |
| frei01 | 0 | 2 | 2 | 0 | 0.33 | I-f |
| frei02 | 2 | 0 | 0 | 2 | 0.33 | I-f |
| holz01 | 16 | 5 | 10 | 11 | 0.11 | Both |
| krem01 | 8 | 1 | 8 | 12 | <0.0001 | Organic/ conventional |
| leto01 | 5 | 0 | 0 | 5 | 0.0080 | Organic/ conventional |
| ober01 | 3 | 2 | 0 | 3 | 0.20 | Both |
| otie01 | 4 | 1 | 5 | 2 | 1.00 | Both |
| rose01 | 0 | 12 | 9 | 0 | <0.0001 | Organic/ conventional |
| saun01 | 5 | 0 | 0 | 10 | 0.0003 | I-f |
| weis01 | 1 | 6 | 3 | 22 | 1.00 | Both |

57

58      **Table S4.** Number of data points grouped by several categories used in the analysis.
59

| (a) Arthropod functional group | | | | |
|---|---|---|---|---|
| **Management scheme** | **Detritivore** | **Herbivore** | **Pollinator** | **Predator** |
| Organic/conventional | 8 | 17 | 20 | 36 |
| In-field plant diversity | 3 | 5 | 13 | 8 |
| **(b) Landscape complexity** | | | | |
| **Management scheme** | **Simple** | **Complex** | **No data** | |
| Organic/conventional | 44 | 30 | 7 | |
| In-field plant diversity | 12 | 17 | 0 | |
| **(c) Biome** | | | | |
| **Management scheme** | **Boreal** | **Mediterranean** | **Temperate** | **Tropical** |
| Organic/conventional | 2 | 14 | 58 | 7 |
| In-field plant diversity | 1 | 9 | 13 | 6 |
| **(d) Cultivation period** | | | | |
| **Management scheme** | **Annual** | **Perennial** | | |
| Organic/conventional | 59 | 22 | | |
| In-field plant diversity | 20 | 9 | | |

60

61 **Table S5.** Correlations between unweighted (log-response ratio) and weighted (Hedges' *d*) effect
62 sizes with various metrics. Weighted metrics could not be calculated at the regional scale (see
63 Methods in main text)
64

| Management scheme | Metric | Pearson's correlation coefficient | *t* | df | *p*-value |
|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 0.66 | 7.88 | 79 | <0.0001 |
| Organic vs. conventional | Local richness | 0.77 | 10.7 | 77 | <0.0001 |
| Organic vs. conventional | Local evenness | 0.70 | 7.99 | 66 | <0.0001 |
| In-field plant diversity | Abundance | 0.90 | 10.7 | 27 | <0.0001 |
| In-field plant diversity | Local richness | 0.81 | 7.26 | 27 | <0.0001 |
| In-field plant diversity | Local evenness | 0.83 | 7.01 | 22 | <0.0001 |

65

66  **Table S6.** Effects of sampling method on effect size (log-response ratio) estimates. ANOVAs testing whether sampling method
67  affected effect sizes were significant in only 4% of cases, which is within the amount expected by chance. Means are average
68  untransformed log-response ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect sizes
69  transformed to percentage change are in parentheses.
70

| Organic vs. conventional | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Functional group** | **Metric** | **N active** | **N passive** | **N both** | **Mean active** | **Mean passive** | **Mean both** | **SE active** | **SE passive** | **SE both** | **F** | ***p-*value** |
| All | Abundance | 32 | 39 | 10 | 0.56 (75%) | 0.19 (21%) | 0.43 (54%) | 0.22 | 0.08 | 0.14 | 1.63 | 0.20 |
| All | Local richness | 32 | 39 | 10 | 0.29 (33%) | 0.14 (15%) | 0.11 (12%) | 0.12 | 0.04 | 0.07 | 1.06 | 0.35 |
| All | Local evenness | 28 | 35 | 10 | -0.07 (-6%) | -0.04 (-4%) | -0.12 (-12%) | 0.07 | 0.05 | 0.05 | 0.22 | 0.80 |
| All | Regional richness | 32 | 39 | 10 | 0.16 (17%) | 0.08 (9%) | 0.07 (7%) | 0.09 | 0.05 | 0.07 | 0.41 | 0.66 |
| All | Regional evenness | 32 | 39 | 10 | -0.22 (-20%) | -0.00 (0%) | 0.04 (5%) | 0.08 | 0.05 | 0.06 | 3.62 | 0.031 |
| Detritivore | Abundance | 3 | 2 | 3 | -0.37 (-31%) | 0.83 (130%) | 0.43 (54%) | 0.51 | 0.63 | 0.29 | 1.70 | 0.27 |
| Detritivore | Local richness | 3 | 2 | 3 | 0.00 (0%) | 0.17 (19%) | -0.09 (-9%) | 0.14 | 0.17 | 0.07 | 1.00 | 0.43 |
| Detritivore | Local evenness | 3 | 1 | 3 | 0.20 (23%) | -0.04 (-4%) | -0.21 (-19%) | 0.13 | NA | 0.07 | 4.15 | 0.11 |
| Detritivore | Regional richness | 3 | 2 | 3 | 0.00 (0%) | -0.38 (-32%) | 0.00 (0%) | 0.11 | 0.31 | 0.16 | 1.26 | 0.36 |
| Detritivore | Regional evenness | 3 | 2 | 3 | 0.22 (24%) | 0.66 (94%) | 0.05 (5%) | 0.11 | 0.96 | 0.08 | 0.60 | 0.58 |
| Herbivore | Abundance | 8 | 6 | 3 | 0.29 (34%) | 0.09 (10%) | 0.39 (47%) | 0.36 | 0.20 | 0.28 | 0.17 | 0.85 |
| Herbivore | Local richness | 8 | 6 | 3 | 0.04 (4%) | 0.16 (17%) | 0.24 (27%) | 0.16 | 0.09 | 0.05 | 0.41 | 0.67 |

| | | N active | N passive | N both | Mean active | Mean passive | Mean both | SE active | SE passive | SE both | F | p-value |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Herbivore | Local evenness | 7 | 4 | 3 | -0.23 (-20%) | -0.17 (-15%) | 0.03 (3%) | 0.21 | 0.19 | 0.09 | 0.32 | 0.73 |
| Herbivore | Regional richness | 8 | 6 | 3 | -0.05 (-5%) | 0.18 (20%) | 0.19 (21%) | 0.11 | 0.10 | 0.05 | 1.53 | 0.25 |
| Herbivore | Regional evenness | 8 | 6 | 3 | -0.25 (-22%) | 0.06 (6%) | 0.12 (13%) | 0.14 | 0.07 | 0.13 | 2.42 | 0.13 |
| Pollinator | Abundance | 12 | 7 | 1 | 0.98 (166%) | -0.10 (-9%) | 1.06 (187%) | 0.45 | 0.26 | NA | 1.56 | 0.24 |
| Pollinator | Local richness | 12 | 7 | 1 | 0.50 (64%) | 0.28 (32%) | 0.41 (51%) | 0.25 | 0.15 | NA | 0.20 | 0.82 |
| Pollinator | Local evenness | 10 | 6 | 1 | 0.02 (2%) | 0.18 (20%) | -0.39 (-33%) | 0.09 | 0.24 | NA | 0.90 | 0.43 |
| Pollinator | Regional richness | 12 | 7 | 1 | 0.26 (30%) | 0.25 (29%) | 0.36 (44%) | 0.19 | 0.11 | NA | 0.02 | 0.98 |
| Pollinator | Regional evenness | 12 | 7 | 1 | -0.16 (-15%) | -0.11 (-10%) | -0.25 (-22%) | 0.15 | 0.11 | NA | 0.06 | 0.94 |
| Predator | Abundance | 9 | 24 | 3 | 0.54 (72%) | 0.24 (28%) | 0.27 (31%) | 0.30 | 0.08 | 0.20 | 0.95 | 0.40 |
| Predator | Local richness | 9 | 24 | 3 | 0.32 (38%) | 0.09 (10%) | 0.08 (8%) | 0.20 | 0.05 | 0.12 | 1.35 | 0.27 |
| Predator | Local evenness | 8 | 24 | 3 | -0.13 (-12%) | -0.08 (-8%) | -0.10 (-9%) | 0.11 | 0.04 | 0.06 | 0.17 | 0.85 |
| Predator | Regional richness | 9 | 24 | 3 | 0.26 (30%) | 0.04 (5%) | -0.07 (-7%) | 0.17 | 0.06 | 0.09 | 1.63 | 0.21 |
| Predator | Regional evenness | 9 | 24 | 3 | -0.42 (-34%) | -0.04 (-4%) | 0.06 (7%) | 0.15 | 0.04 | 0.10 | 7.14 | 0.003 |
| **In-field plant diversity** | | | | | | | | | | | | |
| **Functional group** | **Metric** | **N active** | **N passive** | **N both** | **Mean active** | **Mean passive** | **Mean both** | **SE active** | **SE passive** | **SE both** | **F** | **p-value** |
| All | Abundance | 13 | 11 | 5 | 0.22 (25%) | 0.20 (22%) | 0.37 (45%) | 0.15 | 0.39 | 0.24 | 0.07 | 0.94 |
| All | Local richness | 13 | 11 | 5 | 0.29 (34%) | 0.09 (10%) | 0.26 (30%) | 0.13 | 0.16 | 0.06 | 0.60 | 0.56 |

| | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| All | Local evenness | 12 | 9 | 5 | -0.03 (-3%) | -0.09 (-9%) | -0.11 (-10%) | 0.07 | 0.11 | 0.05 | 0.25 | 0.78 |
| All | Regional richness | 13 | 11 | 5 | 0.25 (28%) | 0.08 (9%) | 0.19 (20%) | 0.13 | 0.14 | 0.09 | 0.44 | 0.65 |
| All | Regional evenness | 13 | 11 | 5 | -0.08 (-8%) | -0.25 (-22%) | -0.04 (-4%) | 0.13 | 0.17 | 0.16 | 0.44 | 0.65 |
| Detritivore | Abundance | 1 | 2 | 0 | 0.03 (3%) | 0.81 (125%) | NA | NA | 1.73 | NA | 0.07 | 0.84 |
| Detritivore | Local richness | 1 | 2 | 0 | -0.07 (-7%) | 0.41 (51%) | NA | NA | 0.45 | NA | 0.39 | 0.65 |
| Detritivore | Local evenness | 0 | 1 | 0 | NA | -0.57 (-44%) | NA | NA | NA | NA | NA | NA |
| Detritivore | Regional richness | 1 | 2 | 0 | 0.41 (50%) | 0.58 (79%) | NA | NA | 0.06 | NA | 2.55 | 0.36 |
| Detritivore | Regional evenness | 1 | 2 | 0 | -0.84 (-57%) | -1.17 (-69%) | NA | NA | 0.33 | NA | 0.32 | 0.67 |
| Herbivore | Abundance | 1 | 3 | 1 | -0.04 (-4%) | 0.30 (35%) | 0.37 (45%) | NA | 0.76 | NA | 0.03 | 0.97 |
| Herbivore | Local richness | 1 | 3 | 1 | 0.12 (13%) | 0.15 (17%) | 0.24 (27%) | NA | 0.45 | NA | 0.01 | 0.99 |
| Herbivore | Local evenness | 1 | 2 | 1 | 0.21 (23%) | -0.17 (-16%) | 0.00 (0%) | NA | 0.19 | NA | 0.71 | 0.64 |
| Herbivore | Regional richness | 1 | 3 | 1 | -0.06 (-6%) | -0.10 (-10%) | 0.15 (16%) | NA | 0.40 | NA | 0.05 | 0.95 |
| Herbivore | Regional evenness | 1 | 3 | 1 | 0.09 (10%) | -0.25 (-22%) | 0.15 (17%) | NA | 0.21 | NA | 0.61 | 0.62 |
| Pollinator | Abundance | 10 | 0 | 3 | 0.37 (46%) | NA | 0.36 (43%) | 0.15 | NA | 0.44 | 0.00 | 0.96 |
| Pollinator | Local richness | 10 | 0 | 3 | 0.40 (49%) | NA | 0.26 (29%) | 0.16 | NA | 0.12 | 0.22 | 0.65 |
| Pollinator | Local evenness | 10 | 0 | 3 | -0.09 (-9%) | NA | -0.17 (-16%) | 0.07 | NA | 0.07 | 0.39 | 0.55 |

| Group | Metric | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Pollinator | Regional richness | 10 | 0 | 3 | 0.28 (32%) | NA | 0.17 (18%) | 0.16 | NA | 0.16 | 0.13 | 0.73 |
| Pollinator | Regional evenness | 10 | 0 | 3 | -0.05 (-5%) | NA | -0.10 (-10%) | 0.15 | NA | 0.28 | 0.03 | 0.88 |
| Predator | Abundance | 1 | 6 | 1 | -0.87 (-58%) | -0.05 (-5%) | 0.40 (50%) | NA | 0.44 | NA | 0.37 | 0.71 |
| Predator | Local richness | 1 | 6 | 1 | -0.22 (-20%) | -0.05 (-5%) | 0.30 (36%) | NA | 0.16 | NA | 0.47 | 0.65 |
| Predator | Local evenness | 1 | 6 | 1 | 0.40 (49%) | 0.01 (1%) | -0.03 (-3%) | NA | 0.13 | NA | 0.71 | 0.54 |
| Predator | Regional richness | 1 | 6 | 1 | 0.08 (8%) | 0.01 (1%) | 0.27 (31%) | NA | 0.13 | NA | 0.30 | 0.75 |
| Predator | Regional evenness | 1 | 6 | 1 | 0.23 (25%) | 0.06 (6%) | -0.04 (-4%) | NA | 0.14 | NA | 0.17 | 0.85 |

71      **Table S7:** Questions investigated in this study, and statistical tests that addressed each one. Q2,
72      Q4, Q7, and Q8 were tested with the same meta-regression.
73

| Question | How tested |
|---|---|
| (Q1) Does diversified farming differentially alter abundance, richness, and evenness? | One sample *t*-tests: Does each metric's mean effect size differ from zero? |
| (Q2) Diversified farming differentially alters local and regional diversity (richness, evenness). | (a) One-sample *t*-tests: Are patterns of difference from zero the same at the local and regional scales? (b) Meta-regression: Does scale affect mean effect size? |
| (Q3) Diversified farming differentially alters abundance and diversity of arthropods in different functional groups | One-sample *t*-tests: Within each functional group (detritivores, herbivores, pollinators, predators), does each metric's mean effect size differ from zero? |
| (Q4) Landscape complexity mediates responses of arthropod communities to diversified farming. | Meta-regression: Do effect sizes differ in simple and complex landscapes? |
| (Q5) Diversified farming differentially affects the abundance and diversity of relatively rare and relatively common taxa. | (a) One-sample *t*-tests: Does each metric's mean effect size for a given rarity category (rare, common) differ from zero? (b) Paired *t*-tests: Within a metric, do mean effect sizes for rare taxa differ from those of common taxa? |
| (Q6) Landscape complexity mediates the degree to which diversified farming differentially affects the abundance and diversity of rare vs. common taxa. | Paired *t*-tests: Within each metric and landscape complexity category (simple, complex), do mean effect sizes for rare taxa differ from those of common taxa? |
| (Q7) A crop's cultivation period (annual, perennial) mediates responses of arthropod communities to diversified farming. | Meta-regression: Do effect sizes differ for crops grown as annuals and perennials? |
| (Q8) Biome mediates responses of arthropod communities to diversified farming. | Meta-regression: Do effect sizes differ among boreal, Mediterranean, temperate, and tropical biomes? |

74
75

76 **Table S8.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
77 diversification impacted overall arthropod communities (pooled across functional groups).
78 Means are average untransformed log-response ratios comparing organic to conventional, or high
79 to low in-field plant diversity, data. Effect sizes transformed to percent change are in
80 parentheses.
81

| Management scheme | Metric | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 81 | 0.36 (45%) | 0.10 | 3.76 | 0.0003 |
| Organic vs. conventional | Local richness | 81 | 0.19 (21%) | 0.05 | 3.75 | 0.0003 |
| Organic vs. conventional | Local evenness | 73 | -0.06 (-6%) | 0.04 | -1.69 | 0.095 |
| Organic vs. conventional | Regional richness | 81 | 0.11 (10%) | 0.04 | 2.52 | 0.014 |
| Organic vs. conventional | Regional evenness | 81 | -0.08 (-9%) | 0.04 | -1.87 | 0.065 |
| In-field plant diversity | Abundance | 29 | 0.24 (27%) | 0.16 | 1.48 | 0.15 |
| In-field plant diversity | Local richness | 29 | 0.21 (23%) | 0.08 | 2.49 | 0.019 |
| In-field plant diversity | Local evenness | 26 | -0.07 (-6%) | 0.05 | -1.31 | 0.20 |
| In-field plant diversity | Regional richness | 29 | 0.17 (19%) | 0.08 | 2.24 | 0.033 |
| In-field plant diversity | Regional evenness | 29 | -0.14 (-13%) | 0.09 | -1.51 | 0.14 |

82
83

84      **Table S9.** Best-fit models, with ΔAICc < 2, and global models explaining arthropod abundance,
85      richness, and evenness in fields managed organically vs. conventionally. *K* is the number of
86      estimated model parameters (fixed plus random effects). Parameters are: F=functional group,
87      D=diversity scale (local, regional), LC=landscape complexity (simple, complex), A=cultivation
88      period (annual, perennial), B=biome. A "*" indicates an interaction and both of its main effects.
89      Detritivores were excluded from meta-regressions due to low sample size.
90

| Abundance | | | | | |
|-----------|------------|---|------|-------|--------|
| **Model ID** | **Parameters** | *K* | **AICc** | **ΔAICc** | **weight** |
| 2 | A | 4 | 178.1 | 0 | 0.40 |
| 6 | F + A | 6 | 178.6 | 0.41 | 0.32 |
| 14 | F + A + LC | 7 | 178.8 | 0.69 | 0.28 |
| Global | F×D + F×LC + D×LC + A + B | 12 | 191.4 | 13.26 | |
| **Richness** | | | | | |
| **Model ID** | **Parameters** | *K* | **AICc** | **ΔAICc** | **weight** |
| 61 | D + F×LC | 9 | 148.1 | 0 | 0.57 |
| 45 | F×LC | 8 | 148.6 | 0.54 | 0.43 |
| Global | F×D + F×LC + D×LC + A + B | 16 | 163.2 | 15.1 | |
| **Evenness** | | | | | |
| **Model ID** | **Parameters** | *K* | **AICc** | **ΔAICc** | **weight** |
| 1 | intercept only | 3 | 82.5 | 0 | 0.52 |
| 17 | D | 4 | 84.0 | 1.5 | 0.25 |
| 2 | A | 4 | 84.0 | 1.5 | 0.24 |
| Global | F×D + F×LC + D×LC + A + B | 16 | 102.7 | 20.2 | |

91    **Table S10.** Regression details for best-fit models listed in Table S7 that explain arthropod abundance, richness, and evenness in fields
92    managed organically vs. conventionally. We significance of fixed effects with likelihood ratio tests (LRTs), and used post-hoc planned
93    contrasts (with *p*-values adjusted via Holm's sequential Bonferroni procedure) to test for (1) differences in effect size among
94    functional groups, and (2) differences in effect size between the local and regional scales within each functional group. Parameters
95    are: F=functional group (h=herbivore, po=pollinator, pr=predator), D=diversity scale (r=regional), LC= landscape complexity
96    (c=complex, s=simple), A=cultivation period (p=perennial), B=biome (b=boreal, M=Mediterranean, te=temperate, tr=tropical). A ":"
97    indicates an interaction. Detritivores were excluded from meta-regressions due to low sample size.
98

| Abundance (detritivores excluded) | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| **Model ID** | **Parameter** | **Coefficient (SE)** | **LRT $\chi^2$** | **LRT df** | **LRT *p*-value** | **Contrast** | **Contrast $\chi^2$** | **Contrast df** | **Contrast *p*-value** |
| 2 | Intercept | 0.54 (0.13) | NA | | | | | | |
| | A, p | -0.50 (0.24) | 4.48 | 1 | 0.034 | | | | |
| 6 | Intercept | 0.41 (0.24) | NA | | | F, h-po | 2.96 | 1 | 0.18 |
| | F, po | 0.52 (0.30) | 4.36 | 2 | 0.11 | F, h-pr | 0.01 | 1 | 0.91 |
| | F, pr | 0.03 (0.28) | | | | F, po-pr | 3.51 | 1 | 0.18 |
| | A, p | -0.62 (0.24) | 6.11 | 1 | 0.014 | | | | |
| 14 | Intercept | 0.09 (0.33) | NA | | | F, h-po | 4.87 | 1 | 0.075 |
| | F, po | 0.75 (0.34) | 6.41 | 2 | 0.041 | F, h-pr | 0.23 | 1 | 0.63 |
| | F, pr | 0.14 (0.28) | | | | F, po-pr | 5.04 | 1 | 0.074 |
| | LC, s | 0.36 (0.25) | 2.22 | 1 | 0.14 | | | | |
| | A, p | -0.57 (0.24) | 5.68 | 1 | 0.017 | | | | |
| **Richness (detritivores excluded)** | | | | | | | | | |
| **Model ID** | **Parameter** | **Coefficient (SE)** | **LRT $\chi^2$** | **LRT df** | **LRT *p*-value** | **Contrast** | **Contrast $\chi^2$** | **Contrast df** | **Contrast *p*-value** |
| 61 | Intercept | -0.46 (0.21) | NA | | | F, h-po | 10.23 | 1 | 0.004 |
| | F, po | 0.88 (0.20) | 18.46 | 4 | 0.001 | F, h-pr | 8.14 | 1 | 0.009 |
| | F, pr | 0.68 (0.20) | | | | F, po-pr | 1.81 | 1 | 0.18 |
| | S, r | -0.09 (0.06) | 2.85 | 1 | 0.092 | F:LC, c-s in h | 6.88 | 1 | 0.026 |
| | LC, s | 0.61 (0.23) | 10.66 | 3 | 0.014 | F:LC, c-s in po | 0.31 | 1 | 1 |
| | F:LC, po | -0.75 (0.32) | 10.64 | 2 | 0.005 | F:LC, c-s | 0.42 | 1 | 1 |

| Model ID | Parameter | Coefficient (SE) | LRT χ² | LRT df | LRT p-value | | | | |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | in pr | | | |
| | F:LC, pr | -0.72 (0.22) | | | | | | | |
| 45 | Intercept | -0.51 (0.21) | NA | | | F, h-po | 10.13 | 1 | 0.004 |
| | F, po | 0.88 (0.20) | 17.95 | 4 | 0.001 | F, h-pr | 7.94 | 1 | 0.010 |
| | F, pr | 0.68 (0.21) | | | | F, po-pr | 1.82 | 1 | 0.18 |
| | LC, s | 0.61 (0.24) | 10.30 | 3 | 0.016 | F:LC, c-s in h | 6.77 | 1 | 0.028 |
| | F:LC, po:s | -0.75 (0.32) | 10.27 | 2 | 0.006 | F:LC, c-s in po | 0.32 | 1 | 1 |
| | F:LC, pr:s | -0.72 (0.23) | | | | F:LC, c-s in pr | 0.41 | 1 | 1 |

**Evenness (detritivores excluded)**

| Model ID | Parameter | Coefficient (SE) | LRT χ² | LRT df | LRT p-value |
|---|---|---|---|---|---|
| 17 | Intercept | -0.08 (0.05) | NA | | |
| | S, r | -0.04 (0.05) | 0.65 | 1 | 0.42 |
| 2 | Intercept | -0.12 (0.06) | NA | | |
| | A, p | 0.07 (0.10) | 0.61 | 1 | 0.43 |

99

100    **Table S11.** Best-fit models, with ΔAICc < 2, and global models explaining arthropod
101    abundance, richness, and evenness in fields managed with high vs. low in-field plant diversity. *K*
102    is the number of estimated model parameters (fixed plus random effects). Parameters are:
103    F=functional group, D=diversity scale (local, regional), LC=landscape complexity (simple,
104    complex), A=cultivation period (annual, perennial), B=biome. A "*" indicates an interaction and
105    both of its main effects.
106

| Abundance | | | | | |
|---|---|---|---|---|---|
| **Model ID** | **Parameters** | ***K*** | **AICc** | **ΔAICc** | **weight** |
| 1 | intercept only | 3 | 70.4 | 0 | 0.67 |
| 2 | A | 4 | 71.7 | 1.4 | 0.33 |
| Global | F×D + F×LC + D×LC + A + B | 14 | 96.7 | 26.3 | |
| **Richness** | | | | | |
| **Model ID** | **Parameters** | ***K*** | **AICc** | **ΔAICc** | **weight** |
| 5 | F | 6 | 42.2 | 0 | 0.36 |
| 45 | F×LC | 10 | 42.2 | 0.04 | 0.36 |
| 7 | F + B | 9 | 42.7 | 0.5 | 0.28 |
| Global | F×D + F×LC + D×LC + A + B | 19 | 54.1 | 11.9 | |
| **Evenness** | | | | | |
| **Model ID** | **Parameters** | ***K*** | **AICc** | **ΔAICc** | **weight** |
| 85 | F×D | 10 | 21.8 | 0 | 1 |
| Global | F×D + F×LC + D×LC + A + B | 19 | 48.5 | 26.7 | |

107 **Table S12.** Regression details for best-fit models listed in Table S9 that explain arthropod abundance, richness, and evenness in fields
108 managed with high vs. low in-field plant diversity. We significance of fixed effects with likelihood ratio tests (LRTs), and used post-
109 hoc planned contrasts (with $p$-values adjusted via Holm's sequential Bonferroni procedure) to test for (1) differences in effect size
110 among functional groups, (2) differences in effect size between the local and regional scales within each functional group, and (3)
111 landscape complexity differences among each pair of functional groups. Parameters are: F=functional group (d=detritivore,
112 h=herbivore, po=pollinator, pr=predator), D=diversity scale (l=local, r=regional), LC= landscape complexity (c=complex, s=simple),
113 A=cultivation period (p=perennial), B=biome (b=boreal, M=Mediterranean, te=temperate, tr=tropical). A ":" indicates an interaction.

**Abundance**

| Model ID | Parameter | Coefficient (SE) | LRT $\chi^2$ | LRT df | LRT $p$-value | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 2 | Intercept | 0.06 (0.20) | NA | | | | | | |
| | A, p | 0.40 (0.36) | 1.33 | 1 | 0.25 | | | | |

**Richness**

| Model ID | Parameter | Coefficient (SE) | LRT $\chi^2$ | LRT df | LRT $p$-value | Contrast | Contrast $\chi^2$ | Contrast df | Contrast $p$-value |
|---|---|---|---|---|---|---|---|---|---|
| 5 | Intercept | 0.25 (0.16) | NA | | | F, d-h | 6.24 | 1 | 0.075 |
| | F, h | -0.30 (0.12) | 9.57 | 3 | 0.023 | F, d-po | 0.10 | 1 | 1 |
| | F, po | 0.06 (0.19) | | | | F, d-pr | 4.13 | 1 | 0.21 |
| | F, pr | -0.24 (0.12) | | | | F, h-po | 4.02 | 1 | 0.21 |
| | | | | | | F, h-pr | 0.31 | 1 | 1 |
| | | | | | | F, po-pr | 3.17 | 1 | 0.23 |
| 45 | Intercept | 0.19 (0.20) | NA | | | F, d-h | 10.37 | 1 | 0.008 |
| | F, h | -0.03 (0.14) | 20.36 | 6 | 0.002 | F, d-po | 0.07 | 1 | 1 |
| | F, po | 0.19 (0.25) | | | | F, d-pr | 7.16 | 1 | 0.037 |
| | F, pr | -0.21 (0.14) | | | | F, h-po | 2.74 | 1 | 0.39 |
| | LC, s | 0.32 (0.34) | 11.00 | 4 | 0.027 | F, h-pr | 0.43 | 1 | 1 |
| | F:LC, h:s | -0.67 (0.23) | 10.57 | 3 | 0.014 | F, po-pr | 1.82 | 1 | 0.53 |
| | F:LC, po:s | -0.49 (0.40) | | | | F:LC, c-s in d | 0.93 | 1 | 1 |
| | F:LC, pr:s | -0.18 (0.23) | | | | F:LC, c-s in h | 1.28 | 1 | 1 |
| | | | | | | F:LC, c-s in po | 0.52 | 1 | 1 |

| Model ID | Parameter | Coefficient (SE) | LRT $\chi^2$ | LRT df | LRT $p$-value | Contrast | Contrast $\chi^2$ | Contrast df | Contrast $p$-value |
|---|---|---|---|---|---|---|---|---|---|
| | | | | | | F:LC, c-s in pr | 0.24 | 1 | 1 |
| 7 | Intercept | 0.30 (0.38) | NA | | | F, d-h | 6.54 | 1 | 0.064 |
| | F, h | -0.31 (0.12) | 11.30 | 3 | 0.010 | F, d-po | 0.29 | 1 | 0.84 |
| | F, po | 0.10 (0.18) | | | | F, d-pr | 3.49 | 1 | 0.19 |
| | F, pr | -0.23 (0.12) | | | | F, h-po | 5.67 | 1 | 0.086 |
| | B, M | 0.17 (0.40) | 7.61 | 3 | 0.054 | F, h-pr | 0.65 | 1 | 0.84 |
| | B, te | -0.28 (0.39) | | | | F, po-pr | 3.93 | 1 | 0.19 |
| | B, tr | 0.09 (0.41) | | | | B, b-M | 0.18 | 1 | 1 |
| | | | | | | B, b-te | 0.51 | 1 | 1 |
| | | | | | | B, b-tr | 0.05 | 1 | 1 |
| | | | | | | B, M-te | 5.54 | 1 | 1 |
| | | | | | | B, M-tr | 0.14 | 1 | 1 |
| | | | | | | B, te-tr | 3.56 | 1 | 1 |

**Richness, boreal data excluded**

| Model ID | Parameter | Coefficient (SE) | LRT $\chi^2$ | LRT df | LRT $p$-value | Contrast | Contrast $\chi^2$ | Contrast df | Contrast $p$-value |
|---|---|---|---|---|---|---|---|---|---|
| 7 | Intercept | 0.47 (0.20) | NA | | | F, d-h | 6.36 | 1 | 0.070 |
| | F, h | -0.31 (0.12) | 10.90 | 3 | 0.012 | F, d-po | 0.29 | 1 | 0.85 |
| | F, po | 0.10 (0.19) | | | | F, d-pr | 3.40 | 1 | 0.20 |
| | F, pr | -0.23 (0.12) | | | | F, h-po | 5.45 | 1 | 0.087 |
| | B, te | -0.45 (0.19) | 7.23 | 2 | 0.027 | F, h-pr | 0.64 | 1 | 0.85 |
| | B, tr | -0.08 (0.22) | | | | F, po-pr | 3.92 | 1 | 0.19 |
| | | | | | | B, M-te | 5.54 | 1 | 0.056 |
| | | | | | | B, M-tr | 0.14 | 1 | 0.71 |
| | | | | | | B, te-tr | 3.56 | 1 | 0.12 |
| 47 | Intercept | 0.41 (0.22) | NA | | | F, d-h | 10.56 | 1 | 0.007 |
| | F, h | -0.03 (0.14) | 21.29 | 6 | 0.002 | F, d-po | 0.01 | 1 | 0.95 |
| | F, po | 0.18 (0.27) | | | | F, d-pr | 6.55 | 1 | 0.052 |
| | F, pr | -0.19 (0.14) | | | | F, h-po | 4.04 | 1 | 0.18 |
| | LC, s | 0.31 (0.36) | 10.55 | 4 | 0.032 | F, h-pr | 0.68 | 1 | 0.82 |
| | B, te | -0.43 (0.21) | 7.07 | 2 | 0.029 | F, po-pr | 2.55 | 1 | 0.33 |

| Model ID | Parameter | Coefficient (SE) | LRT χ² | LRT df | LRT p-value | Contrast | Contrast χ² | Contrast df | Contrast p-value |
|---|---|---|---|---|---|---|---|---|---|
| | B, tr | -0.05 (0.29) | | | | B, M-te | 4.39 | 1 | 0.11 |
| | F:LC, h:s | -0.69 (0.23) | 10.30 | 3 | 0.016 | B, M-tr | 0.03 | 1 | 0.86 |
| | F:LC, po:s | -0.39 (0.47) | | | | B, te-tr | 2.40 | 1 | 0.24 |
| | F:LC, pr:s | -0.22 (0.23) | | | | F:LC, c-s in d | 0.73 | 1 | 1 |
| | | | | | | F:LC, c-s in h | 1.23 | 1 | 1 |
| | | | | | | F:LC, c-s in po | 0.12 | 1 | 1 |
| | | | | | | F:LC, c-s in pr | 0.08 | 1 | 1 |
| 5 | Intercept | 0.24 (0.17) | NA | | | F, d-h | 6.04 | 1 | 0.084 |
| | F, h | -0.30 (0.12) | 9.21 | 3 | 0.027 | F, d-po | 0.12 | 1 | 1 |
| | F, po | 0.07 (0.20) | | | | F, d-pr | 4.02 | 1 | 0.22 |
| | F, pr | -0.25 (0.12) | | | | F, h-po | 3.84 | 1 | 0.22 |
| | | | | | | F, h-pr | 0.29 | 1 | 1 |
| | | | | | | F, po-pr | 2.98 | 1 | 0.25 |
| **Evenness** | | | | | | | | | |
| **Model ID** | **Parameter** | **Coefficient (SE)** | **LRT χ²** | **LRT df** | **LRT p-value** | **Contrast** | **Contrast χ²** | **Contrast df** | **Contrast p-value** |
| 85 | Intercept | -0.08 (0.21) | NA | | | F, d-h | 17.99 | 1 | 0.0001 |
| | F, h | 0.14 (0.21) | 46.79 | 6 | <0.0001 | F, d-po | 6.45 | 1 | 0.045 |
| | F, po | -0.04 (0.23) | | | | F, d-pr | 21.60 | 1 | <0.0001 |
| | F, pr | 0.13 (0.20) | | | | F, h-po | 0.59 | 1 | 0.89 |
| | S, r | -0.88 (0.21) | 16.44 | 4 | 0.003 | F, h-pr | 0.18 | 1 | 0.89 |
| | F:S, h:r | 0.79 (0.24) | 16.13 | 3 | 0.001 | F, po-pr | 1.21 | 1 | 0.81 |
| | F:S, po:r | 0.92 (0.22) | | | | F:S, l-r in d | 17.44 | 1 | 0.0001 |
| | F:S, pr:r | 0.89 (0.23) | | | | F:S, l-r in h | 0.55 | 1 | 1 |
| | | | | | | F:S, l-r in po | 0.44 | 1 | 1 |
| | | | | | | F:S, l-r in pr | 0.01 | 1 | 1 |

114  **Table S13.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
115  diversification impacted overall arthropod communities (pooled across functional groups) for
116  rare and common taxa. We classified taxa as common if their relative abundance was at least 5%
117  of the total community; other species were categorized as rare. Means are average untransformed
118  log-response ratios comparing organic to conventional, or high to low in-field plant diversity,
119  data. Effect sizes transformed to percent change are in parentheses.
120

| Management scheme | Metric | Relative abundance category | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | Rare | 77 | 0.44 (55%) | 0.45 | 4.16 | <0.0001 |
| Organic vs. conventional | Abundance | Common | 82 | 0.37 (45%) | 0.51 | 3.64 | <0.0001 |
| Organic vs. conventional | Local richness | Rare | 77 | 0.24 (27%) | 0.38 | 3.29 | 0.002 |
| Organic vs. conventional | Local richness | Common | 82 | 0.13 (14%) | 0.31 | 2.75 | 0.007 |
| Organic vs. conventional | Regional richness | Rare | 73 | 0.12 (12%) | 0.31 | 2.52 | 0.014 |
| Organic vs. conventional | Regional richness | Common | 78 | 0.05 (6%) | 0.29 | 1.80 | 0.076 |
| In-field plant diversity | Abundance | Rare | 25 | 0.23 (25%) | 1.31 | 1.33 | 0.19 |
| In-field plant diversity | Abundance | Common | 30 | 0.31 (37%) | 1.10 | 1.79 | 0.084 |
| In-field plant diversity | Local richness | Rare | 25 | 0.33 (39%) | 0.68 | 2.24 | 0.035 |
| In-field plant diversity | Local richness | Common | 30 | 0.13 (14%) | 0.31 | 2.17 | 0.038 |
| In-field plant diversity | Regional richness | Rare | 24 | 0.24 (28%) | 0.69 | 1.89 | 0.071 |
| In-field plant diversity | Regional richness | Common | 25 | 0.04 (4%) | 0.18 | 1.45 | 0.16 |

121

122   **Table S14.** Results of paired *t*-tests testing whether organic farming and in-field plant diversification impacted arthropod abundance
123   and richness differentially for rare and common taxa. Means are average untransformed log-response ratios comparing organic to
124   conventional, or high to low in-field plant diversity, data. Effect sizes transformed to percent change are in parentheses.
125

| Management scheme | Metric | N common taxa | Mean common taxa | SE common taxa | N rare taxa | Mean rare taxa | SE rare taxa | *t* | *p*-value |
|---|---|---|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 82 | 0.37 (45%) | 0.10 | 77 | 0.44 (55%) | 0.11 | -0.76 | 0.45 |
| Organic vs. conventional | Local richness | 82 | 0.13 (14%) | 0.05 | 77 | 0.24 (27%) | 0.07 | -2.40 | 0.019 |
| Organic vs. conventional | Regional richness | 78 | 0.05 (6%) | 0.03 | 73 | 0.12 (12%) | 0.05 | -1.63 | 0.11 |
| In-field plant diversity | Abundance | 30 | 0.31 (37%) | 0.17 | 25 | 0.23 (25%) | 0.17 | 1.02 | 0.32 |
| In-field plant diversity | Local richness | 30 | 0.13 (14%) | 0.06 | 25 | 0.33 (39%) | 0.15 | -1.61 | 0.12 |
| In-field plant diversity | Regional richness | 25 | 0.04 (4%) | 0.02 | 24 | 0.24 (28%) | 0.13 | -1.48 | 0.15 |

126

127 **Table S15.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
128 diversification impacted pollinator communities. Means are average untransformed log-response
129 ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect
130 sizes transformed to percent change are in parentheses.
131

| Management scheme | Metric | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 20 | 0.61 (90%) | 0.30 | 2.01 | 0.058 |
| Organic vs. conventional | Local richness | 20 | 0.42 (55%) | 0.16 | 2.68 | 0.015 |
| Organic vs. conventional | Local evenness | 17 | 0.05 (5%) | 0.10 | 0.52 | 0.61 |
| Organic vs. conventional | Regional richness | 20 | 0.27 (32%) | 0.12 | 2.25 | 0.036 |
| Organic vs. conventional | Regional evenness | 20 | -0.15 (-15%) | 0.10 | -1.58 | 0.13 |
| In-field plant diversity | Abundance | 13 | 0.37 (45%) | 0.14 | 2.62 | 0.023 |
| In-field plant diversity | Local richness | 13 | 0.36 (44%) | 0.12 | 2.97 | 0.012 |
| In-field plant diversity | Local evenness | 13 | -0.11 (-11%) | 0.05 | -2.07 | 0.061 |
| In-field plant diversity | Regional richness | 13 | 0.25 (29%) | 0.13 | 2.01 | 0.068 |
| In-field plant diversity | Regional evenness | 13 | -0.07 (-6%) | 0.13 | -0.51 | 0.62 |

132

133    **Table S16.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
134    diversification impacted predator communities. Means are average untransformed log-response
135    ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect
136    sizes transformed to percent change are in parentheses.
137

| Management scheme | Metric | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 36 | 0.32 (39%) | 0.09 | 3.41 | 0.0020 |
| Organic vs. conventional | Local richness | 36 | 0.15 (14%) | 0.06 | 2.42 | 0.021 |
| Organic vs. conventional | Local evenness | 35 | -0.09 (-9%) | 0.03 | -2.69 | 0.011 |
| Organic vs. conventional | Regional richness | 36 | 0.09 (6%) | 0.06 | 1.50 | 0.14 |
| Organic vs. conventional | Regional evenness | 36 | -0.12 (-14%) | 0.05 | -2.35 | 0.024 |
| In-field plant diversity | Abundance | 8 | -0.10 (-10%) | 0.34 | -0.29 | 0.78 |
| In-field plant diversity | Local richness | 8 | -0.03 (-3%) | 0.13 | -0.19 | 0.85 |
| In-field plant diversity | Local evenness | 8 | 0.06 (6%) | 0.10 | 0.54 | 0.61 |
| In-field plant diversity | Regional richness | 8 | 0.05 (5%) | 0.10 | 0.51 | 0.63 |
| In-field plant diversity | Regional evenness | 8 | 0.07 (7%) | 0.10 | 0.63 | 0.55 |

138

139 **Table S17.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
140 diversification impacted herbivore communities. Means are average untransformed log-response
141 ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect
142 sizes transformed to percent change are in parentheses.
143

| Management scheme | Metric | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 17 | 0.24 (23%) | 0.18 | 1.30 | 0.21 |
| Organic vs. conventional | Local richness | 17 | 0.12 (10%) | 0.08 | 1.44 | 0.17 |
| Organic vs. conventional | Local evenness | 14 | -0.16 (-14%) | 0.12 | -1.33 | 0.21 |
| Organic vs. conventional | Regional richness | 17 | 0.07 (5%) | 0.07 | 1.06 | 0.30 |
| Organic vs. conventional | Regional evenness | 17 | -0.07 (-7%) | 0.08 | -0.89 | 0.39 |
| In-field plant diversity | Abundance | 5 | 0.25 (28%) | 0.42 | 0.58 | 0.59 |
| In-field plant diversity | Local richness | 5 | 0.17 (18%) | 0.25 | 0.67 | 0.54 |
| In-field plant diversity | Local evenness | 4 | -0.04 (-4%) | 0.12 | -0.30 | 0.78 |
| In-field plant diversity | Regional richness | 5 | -0.04 (-4%) | 0.23 | -0.20 | 0.85 |
| In-field plant diversity | Regional evenness | 5 | -0.10 (-10%) | 0.15 | 0.68 | 0.53 |

144

145    **Table S18.** Results of one-sample *t*-tests testing whether organic farming and in-field plant
146    diversification impacted detritivore communities. Means are average untransformed log-response
147    ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect
148    sizes transformed to percent change are in parentheses.
149

| Management scheme | Metric | N | Mean | SE | *t* | *p*-value |
|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | 8 | 0.23 (26%) | 0.29 | 0.79 | 0.46 |
| Organic vs. conventional | Local richness | 8 | 0.01 (1%) | 0.07 | 0.15 | 0.89 |
| Organic vs. conventional | Local evenness | 7 | -0.01 (-1%) | 0.09 | -0.06 | 0.95 |
| Organic vs. conventional | Regional richness | 8 | -0.10 (-9%) | 0.11 | -0.91 | 0.39 |
| Organic vs. conventional | Regional evenness | 8 | 0.26 (30%) | 0.21 | 1.28 | 0.24 |
| In-field plant diversity | Abundance | 3 | 0.55 (74%) | 1.03 | 0.54 | 0.65 |
| In-field plant diversity | Local richness | 3 | 0.25 (28%) | 0.31 | 0.82 | 0.50 |
| In-field plant diversity | Local evenness | 1 | -0.57 (-44%) | NA | NA | NA |
| In-field plant diversity | Regional richness | 3 | 0.52 (69%) | 0.07 | 7.51 | 0.017 |
| In-field plant diversity | Regional evenness | 3 | -1.06 (-65%) | 0.22 | -4.80 | 0.041 |

150

151   **Table S19.** Results of paired *t*-tests testing whether organic farming and in-field plant diversification impacted arthropod abundance
152   and richness differentially for rare and common taxa, in simple and complex landscapes. Means are average untransformed log-
153   response ratios comparing organic to conventional, or high to low in-field plant diversity, data. Effect sizes transformed to percent
154   change are in parentheses.
155

| Management scheme | Metric | Landscape complexity | N common taxa | Mean common taxa | SE common taxa | N rare taxa | Mean rare taxa | SE rare taxa | *t* | *p*-value |
|---|---|---|---|---|---|---|---|---|---|---|
| Organic vs. conventional | Abundance | Simple | 45 | 0.45 (57%) | 0.12 | 43 | 0.36 (44%) | 0.11 | 0.51 | 0.61 |
| Organic vs. conventional | Abundance | Complex | 30 | 0.28 (33%) | 0.21 | 28 | 0.58 (78%) | 0.24 | -1.90 | 0.068 |
| Organic vs. conventional | Local richness | Simple | 45 | 0.09 (10%) | 0.05 | 43 | 0.15 (16%) | 0.07 | -0.88 | 0.39 |
| Organic vs. conventional | Local richness | Complex | 30 | 0.19 (21%) | 0.10 | 28 | 0.36 (44%) | 0.16 | -2.35 | 0.027 |
| Organic vs. conventional | Regional richness | Simple | 42 | 0.05 (6%) | 0.04 | 41 | 0.06 (6%) | 0.06 | 0.10 | 0.92 |
| Organic vs. conventional | Regional richness | Complex | 29 | 0.04 (4%) | 0.04 | 26 | 0.16 (17%) | 0.07 | -2.33 | 0.028 |
| In-field plant diversity | Abundance | Simple | 13 | 0.24 (27%) | 0.22 | 10 | 0.08 (8%) | 0.07 | 1.58 | 0.15 |
| In-field plant diversity | Abundance | Complex | 17 | 0.37 (45%) | 0.27 | 15 | 0.33 (39%) | 0.28 | 0.05 | 0.96 |
| In-field plant diversity | Local richness | Simple | 13 | 0.09 (10%) | 0.08 | 10 | 0.05 (5%) | 0.10 | 1.00 | 0.35 |
| In-field plant diversity | Local richness | Complex | 17 | 0.16 (18%) | 0.09 | 15 | 0.52 (68%) | 0.23 | -2.22 | 0.044 |
| In-field plant diversity | Regional richness | Simple | 10 | 0.06 (6%) | 0.06 | 10 | 0.02 (2%) | 0.09 | -0.04 | 0.97 |
| In-field plant diversity | Regional richness | Complex | 15 | 0.02 (2%) | 0.01 | 14 | 0.40 (50%) | 0.20 | -1.59 | 0.14 |

156

157   **Fig. S1.** Data structure and major factors used in the meta-analysis. Each study consisted of a
158   collection of fields (white rectangles, not to scale) situated in simple or complex landscapes. We
159   classified each field as having low or high in-field plant diversity, or being managed organically
160   or conventionally (not shown). Within each study, we divided sampled taxa by functional group
161   (detritivore, herbivore, pollinator, predator). For each sub-group, we calculated local abundance
162   and diversity from field-level taxon pools, and regional diversity from the regional pool.
163
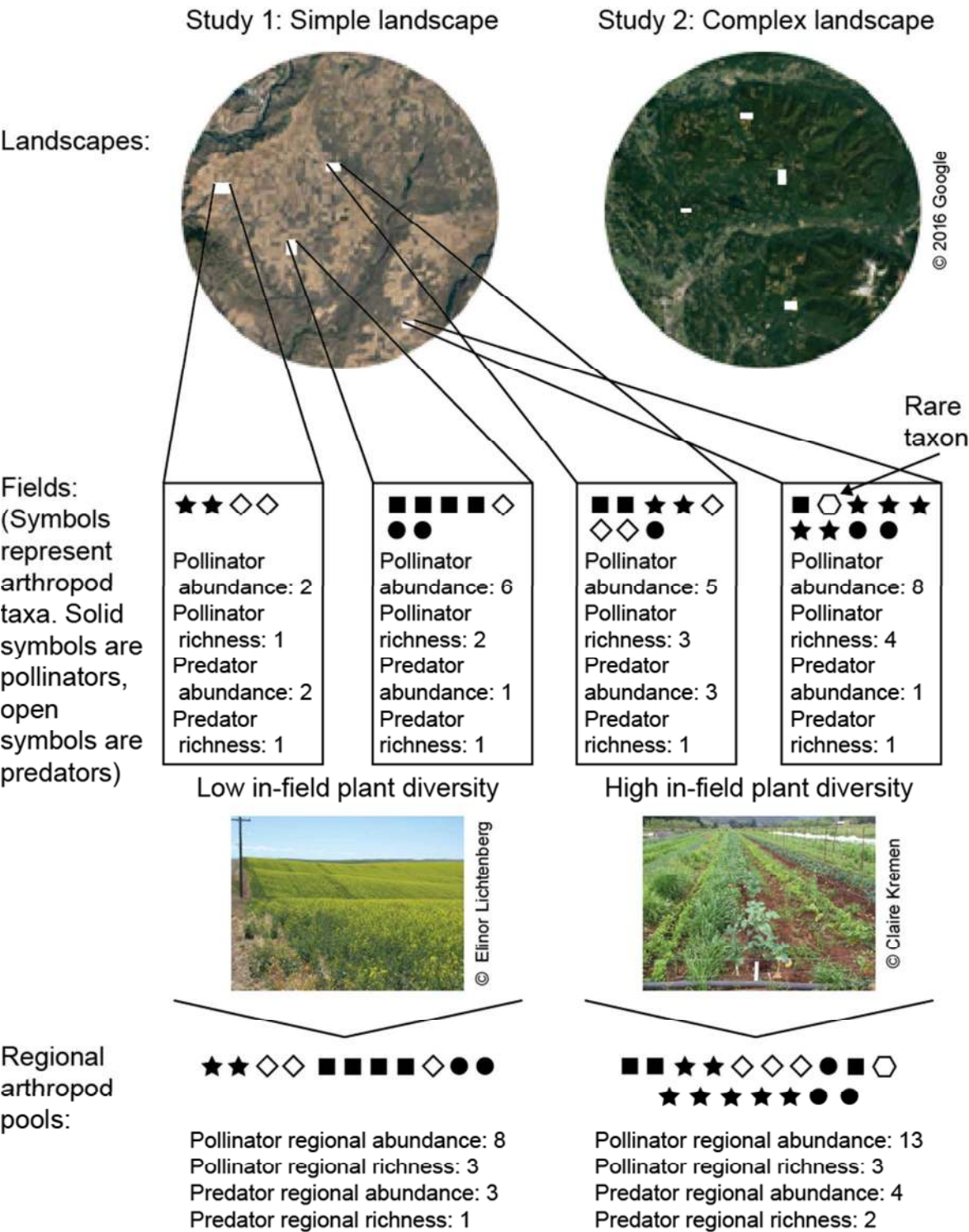
165    **Fig. S2.** Map of study sites.
166



167
168

169    **Figure S3.** Funnel plots for studies assessing organic vs. conventional farming. All plots are
170    sufficiently symmetrical about their mean (visually assessed) to indicate no publication bias.
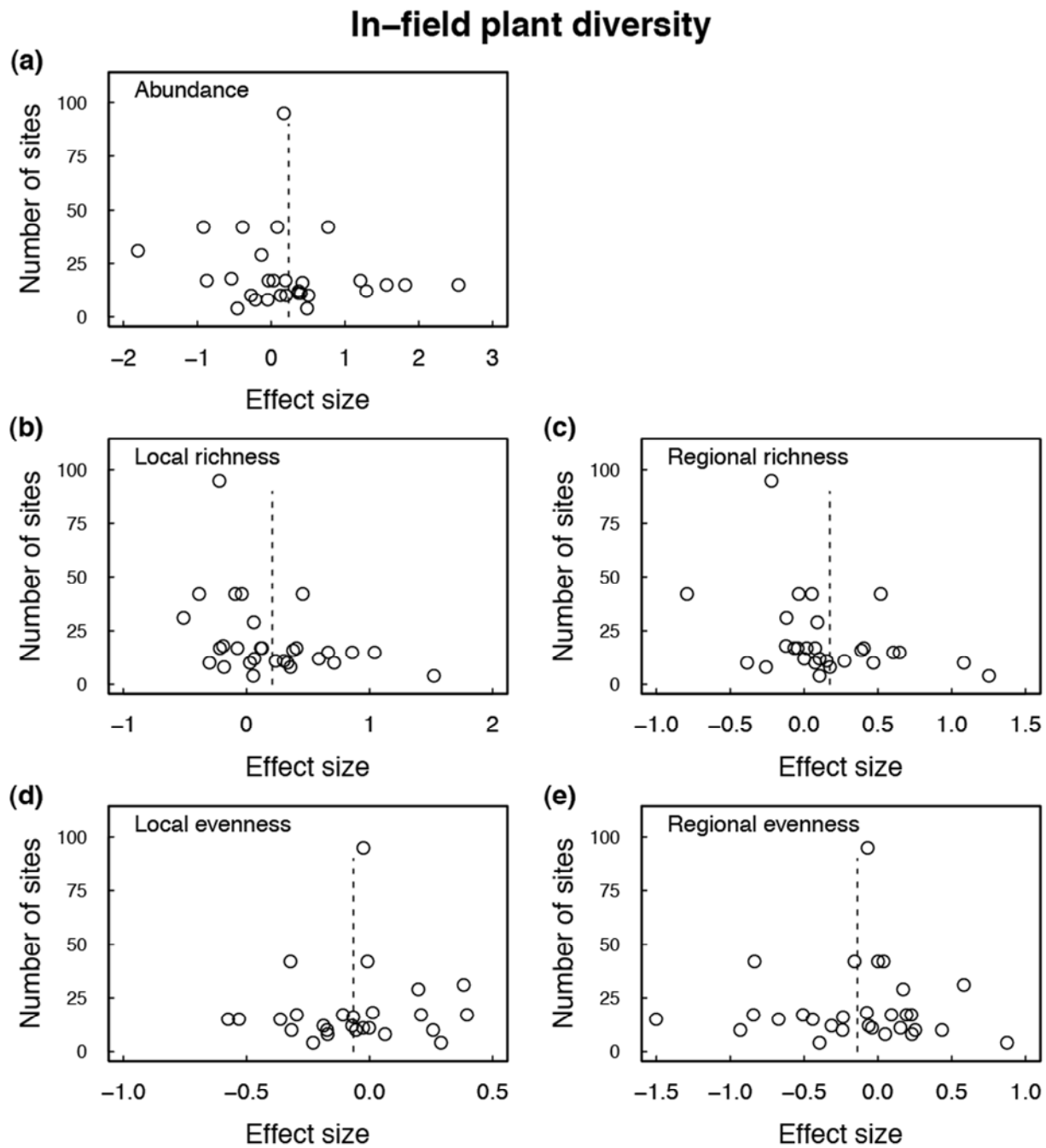171    Effect sizes are log-response ratios.
172



173
174
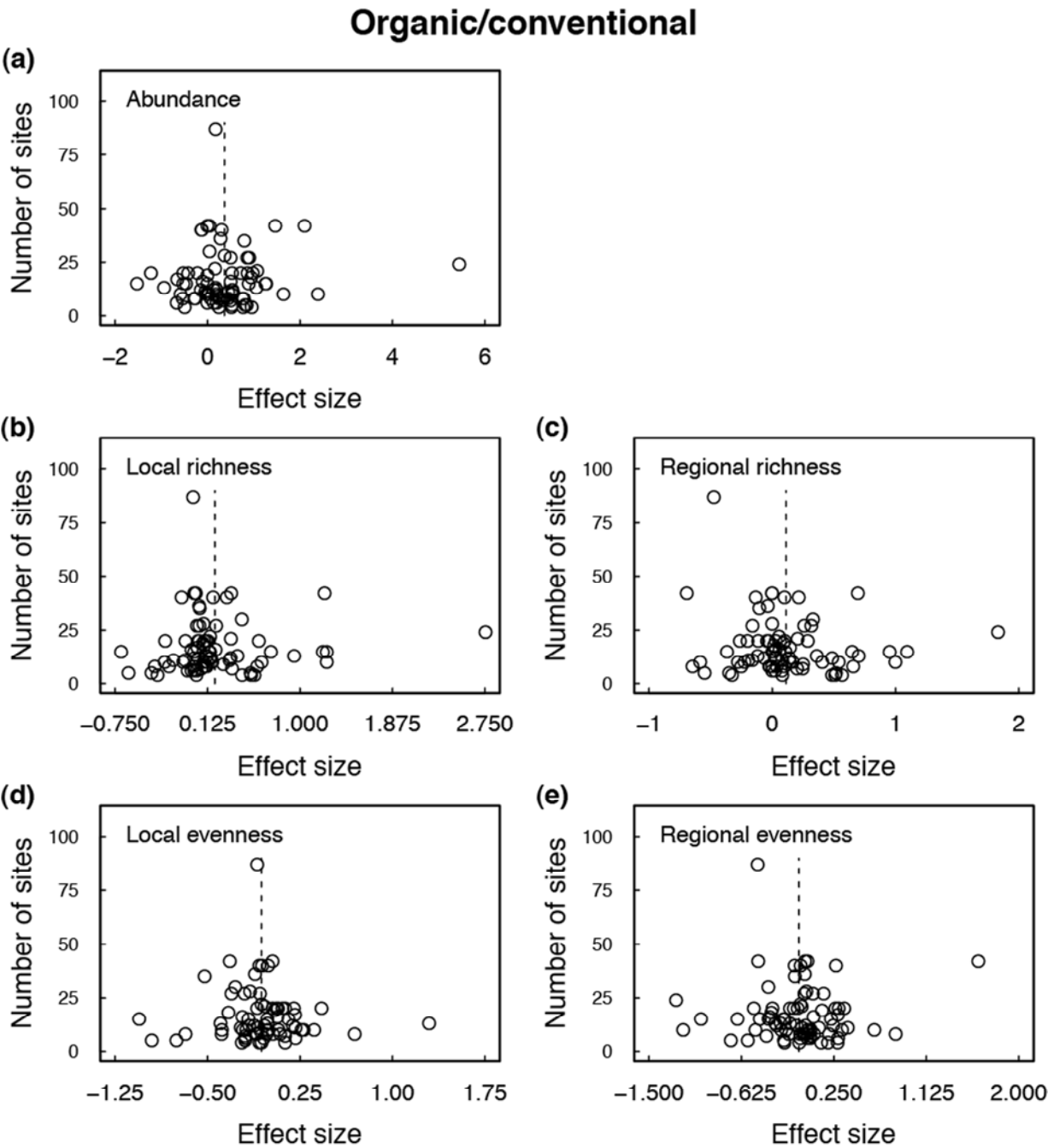
175   **Figure S4.** Funnel plots for studies assessing in-field plant diversification. All plots are
176   sufficiently symmetrical about their mean (visually assessed) to indicate no publication bias.
177   Effect sizes are log-response ratios.
178



## In–field plant diversity

179
180

181    **Figure S5.** Diversity effects (log-response ratios) strongly correlated at the local and regional
182    scales for both richness (Pearson's correlation: $r = 0.87$, $t_{108} = 18.41$, $p < 0.0001$) and evenness ($r$
183    $= 0.81$, $t_{97} = 5.83$, $p < 0.0001$).
184



185
186
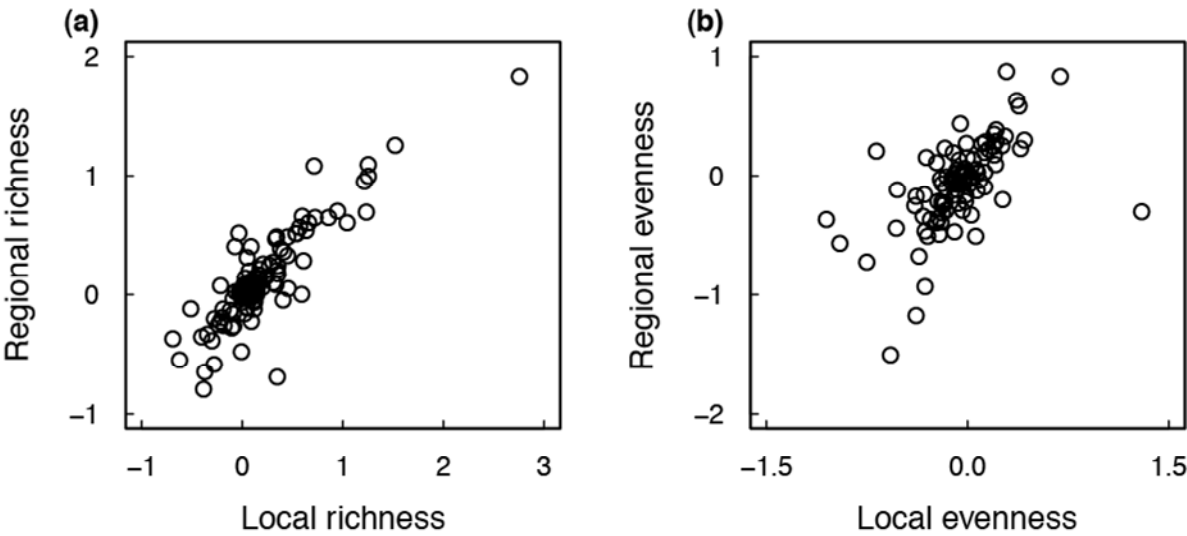
187    **Figure S6.** Effects of farm management schemes on abundance (a, b) and richness (c, d) of
188    common vs. rare taxa. Mean log-response ratios (±SE) of (left column) adopting organic farming
189    and (right column) promoting in-field plant diversity. A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$)
190    above a mean denotes a significant difference from zero (determined via one-sample $t$-tests),
191    while one below a pair of means indicates a significant difference between rare and common
192    taxa (determined via paired $t$-tests).

193



194
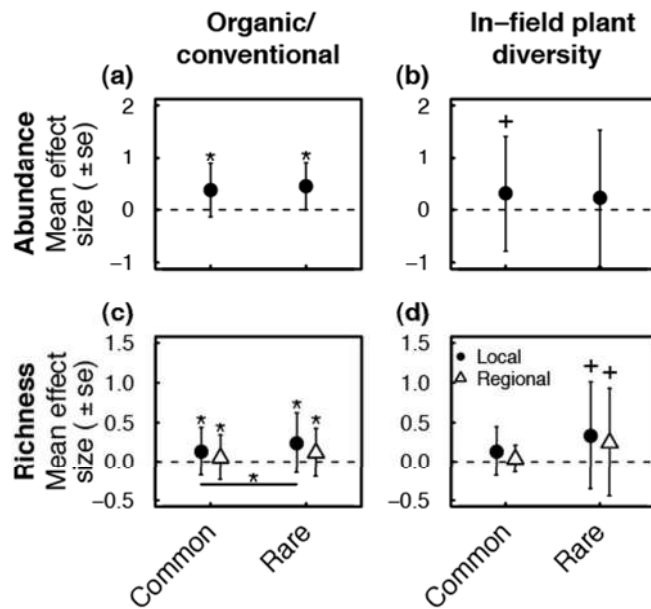
**Table S1.** Data holders and studies participating. We were unable to categorize landscape complexity when we obtained data directly from published articles that lacked GPS coordinates of sampling locations or information on natural habitat surrounding fields (Study IDs drit01, febe01, hesl01, hokk01, and weib01). These studies were excluded from meta-regressions.

| Study ID | Reference or data holder | Crop(s) | Study location | Functional group(s) | Management scheme(s) | # sites (o=organic/ conventional, i-f=in-field plant diversity) | Year(s) |
|---|---|---|---|---|---|---|---|
| arms01 | (Armstrong, 1995) | potato | Scotland | predators | organic/ conventional | 4 | 1992 |
| bata01 | (Batáry *et al.*, 2012) | wheat | Germany | predators | organic/ conventional | 18 | 2008 |
| benj01 | (Cariveau *et al.*, 2013) | blueberry | USA | pollinators | in-field plant diversity | 16 | 2012 |
| bere01 | (Winqvist *et al.*, 2011) | wheat | Netherlands | predators | organic/ conventional | 35 | 2007 |
| bomm01 | (Winqvist *et al.*, 2011) | wheat | Sweden | predators | organic/ conventional, in-field plant diversity | 95 | 2007 |
| bosq01 | Bosque-Perez, Nilsa; Ramos, Mariangie | coffee | Costa Rica | herbivores | organic/ conventional, in-field plant diversity | 18 (o), 19 (i-f) | 2005 |
| carv01 | (Carvalheiro *et al.*, 2010, 2012) | mango | South Africa | herbivores, pollinators, predators | organic/ conventional | 15 | 2009 |

1

| chap01 | (Chaplin-Kramer *et al.*, 2013) | broccoli | USA | detritivores, herbivores, predators | in-field plant diversity | 17 | 2008 |
|---|---|---|---|---|---|---|---|
| clou01 | (Clough *et al.*, 2005, 2007a, 2007b) | wheat | Germany | detritivores, herbivores, predators | organic/ conventional, in-field plant diversity | 42 (o), 17 (i-f) | 2003 |
| conn01 | (Connelly *et al.*, 2015) | strawberry | USA | pollinators | organic/ conventional | 13 | 2012 |
| danf01 | (Russo *et al.*, 2015) | apple | USA | pollinators | organic/ conventional, in-field plant diversity | 10 | 2009 |
| diek01 | (Diekötter *et al.*, 2010) | wheat | Germany | detritivores, herbivores, predators | organic/ conventional | 12 | 2007 |
| drit01 | (Dritschilo & Erwin, 1982) | corn | USA | predators | organic/ conventional | 8 | late 1970s? |
| eige01 | Eigenbrode, Sanford | coffee | Costa Rica | predators | organic/ conventional | 6 | 2001 |
| ekro01 | (Ekroos *et al.*, 2010) | various grains (combined) | Finland | predators | organic/ conventional, in-field plant diversity | 28 (o), 29 (i-f) | 1998 |
| febe01 | (Feber *et al.*, 1998) | wheat | England | predators | organic/ conventional | 6 | 1995 |
| frei01 | Freitas, Breno | acerola | Brazil | pollinators | in-field plant diversity | 4 | 2010 |

2

| frei02 | Freitas, Breno | cotton | Brazil | pollinators | in-field plant diversity | 4 | 2010 |
|--------|----------------|--------|--------|-------------|--------------------------|---|------|
| fuku01 | (Fukuda *et al.*, 2011) | pasture | New Zealand | detritivores, herbivores, predators | organic/ conventional | 20 | 2009 |
| gain01 | Gaines, Hannah; Gratton, Claudio | cranberry | USA | pollinators | organic/ conventional | 15 | 2008 |
| hesl01 | (Hesler *et al.*, 1993) | rice | USA | herbivores, predators | organic/ conventional | 6 | 1988 |
| hokk01 | (Hokkanen & Holopainen, 1986) | cabbage | Germany | herbivores, predators | organic/ conventional | 4 | 1982 |
| holz01 | (Holzschuh *et al.*, 2007) | wheat | Germany | pollinators | organic/ conventional, in-field plant diversity | 42 | 2003 |
| isaa01 | (Isaacs & Kirk, 2010) | blueberry | USA | pollinators | in-field plant diversity | 12 | 2008 |
| isai01 | (Isaia *et al.*, 2006) | grape | Italy | predators | organic/ conventional | 5 | 2003 |
| jha01 | (Jha & Vandermeer, 2010) | coffee | Mexico | pollinators | organic/ conventional | 7 | 2006 |
| jona01 | (Jonason *et al.*, 2013) | various grains (combined) | Sweden | herbivores, predators | organic/ conventional | 36 | 2011 |

3

| | | | | | | | |
|---|---|---|---|---|---|---|---|
| jone01 | (Jones *et al.*, In press, In pressb; Mills *et al.*, In press) | apple | USA | herbivores, pollinators, predators | organic/ conventional | 8 | 2011 |
| klat01 | Klatt, Björn; Tscharntke, Teja | strawberry | Germany | pollinators | in-field plant diversity | 8 | 2010 |
| klei01 | Brittain, Claire; Klein, Alexandra | almond | USA | pollinators | organic/ conventional | 13 | 2009 |
| krau01 | (Krauss *et al.*, 2011) | triticale | Germany | pollinators | organic/ conventional | 24 | 2008 |
| krem01 | (Kremen *et al.*, 2002, 2004) | watermelon | USA | pollinators | organic/ conventional | 21 | 2000 |
| leto01 | (Drinkwater *et al.*, 1995; Letourneau & Goldstein, 2001; Letourneau & Bothwell, 2007; Letourneau *et al.*, 2012, 2015) | broccoli, brussel sprouts | USA | predators | organic/ conventional, in-field plant diversity | 10 | 2006 |
| mall01 | (Mallinger *et al.*, 2015) | apple | USA | pollinators | organic/ conventional | 17 | 2012 |
| mart01 | (Martin *et al.*, 2016) | potato, daikon radish, rice, soybean | South Korea | predators | organic/ conventional | 7 (radish), 8 (other crops) | 2009 |

4

| memm01 | (Gibson *et al.*, 2007; Macfadyen *et al.*, 2009a, 2009b, 2011a, 2011b) | grains, brassicas, legumes | England | herbivores, predators | organic/ conventional | 20 (grains), 5 (brassicas), 10 (legumes) | 2005 (grains, legumes), 2006 (brassicas) |
|--------|-----------|-----------|---------|-----------|-----------|-----------|-----------|
| mora01 | (Morandin & Winston, 2005, 2006) | canola | Canada | pollinators | organic/ conventional | 16 | 2002 |
| neam01 | Elle, Elizabeth; Neame, Lisa | winter squash | Canada | pollinators | organic/ conventional | 9 | 2010 |
| ober01 | (Öberg, 2007; Öberg *et al.*, 2007) | various grains (combined) | Sweden | predators | organic/ conventional, in-field plant diversity | 8 | 2003 (i-f), 2004 (o) |
| otie01 | (Otieno *et al.*, 2015) | pigeonpea | Kenya | pollinators | organic/ conventional, in-field plant diversity | 12 | 2009 |
| pfif01 | (Pfiffner & Luka, 2003) | various grains (combined) | Switzerland | predators | organic/ conventional | 12 | 1996-8 |
| poco01 | (Pocock & Jennings, 2008) | various grains (combined) | England | detritivores, herbivores, predators | organic/ conventional | 40 | 2003 |
| ponc01 | (Ponce *et al.*, 2011) | wheat, barley | Spain | detritivores, herbivores, predators | organic/ conventional | 27 (wheat), 11 (barley) | 2008 |
| pott01 | (Carré *et al.*, 2009) | field bean | England | pollinators | in-field plant diversity | 10 | 2005 |

5

| pove01 | (Poveda *et al.*, 2012); Martinez, Eliana | potato | Colombia | herbivores, predators | in-field plant diversity | 11 | 2007 |
|--------|-------------------------------------------|--------|----------|-----------------------|--------------------------|----|------|
| rose01 | (de Valpine & Rosenheim, 2008) | cotton | USA | herbivores, predators | organic/ conventional | 15 | 1993 |
| rund01 | (Bommarco *et al.*, 2012) | red clover | Sweden | pollinators | in-field plant diversity | 17 | 2010 |
| sard01 | (Sardiñas & Kremen, 2015) | sunflower | USA | pollinators | in-field plant diversity | 10 | 2011 |
| saun01 | (Saunders & Luck, 2013) | almond | Australia | detritivores, herbivores, predators | in-field plant diversity | 15 | 2010 |
| scho01 | (Schon *et al.*, 2011) | pasture | New Zealand | detritivores, herbivores, predators | organic/ conventional | 10 | 2007 |
| scil01 | Sciligo, Amber | strawberry | USA | pollinators | in-field plant diversity | 17 | 2012 |
| sidh01 | (Sidhu, 2013) | squash | USA | pollinators | organic/ conventional | 8 | 2011 |
| snyd01 | Crowder, David; Snyder, William | potato | USA | detritivores, herbivores, predators | organic/ conventional | 20 | 2010 |
| vese01 | (Veselý & Šarapatka, 2008) | wheat, barley | Czech Republic | predators | organic/ conventional | 4 (wheat), 4 (barley) | 2001 (wheat), 2005 (barley) |

6

| weib01 | (Weibull *et al.*, 2000) | cereals, clovers, grasses (combined) | Sweden | pollinators | organic/ conventional | 16 | 1997-8 |
|---|---|---|---|---|---|---|---|
| weis01 | (Winqvist *et al.*, 2011) | wheat | Germany | predators | organic/ conventional, in-field plant diversity | 30 (o), 31 (i-f) | 2007 |
| will01 | Williams, Neal | watermelon | USA | pollinators | in-field plant diversity | 10 | 2010 |
| wils01 | (Tuell *et al.*, 2009) | blueberry | USA | pollinators | organic/ conventional | 15 | 2005 |
| winf01 | (Winfree *et al.*, 2007, 2008; Lonsdorf *et al.*, 2009; Rader *et al.*, 2013) | watermelon | USA | pollinators | organic/ conventional | 10 | 2010 |
| winf02 | (Winfree *et al.*, 2008) | pepper, tomato | USA | pollinators | organic/ conventional | 22 (pepper), 13 (tomato) | 2004 (pepper), 2005 (tomato) |

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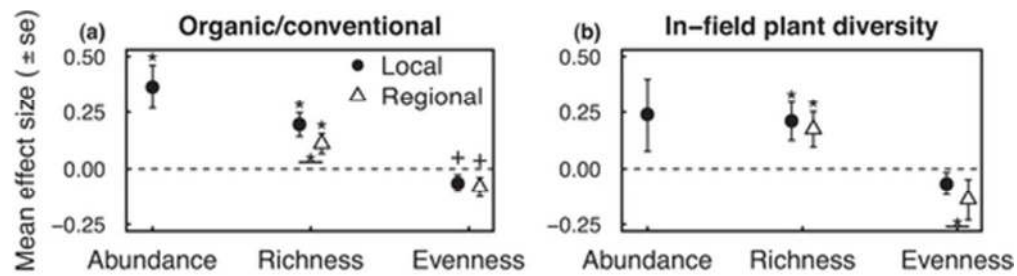
11

Effects of farm management schemes on arthropod abundance, local diversity, and regional diversity. Values shown are for the entire arthropod community, and represent the mean log-response ratio (± SE) of (a) adopting organic farming and (b) promoting in-field plant diversity on abundance, richness, and evenness. A "*" ($p < 0.05$) or "+" ($0.05 \le p < 0.1$) above a mean denotes a significant difference from zero (determined via one-sample t-tests; statistical details in Table S8), while one below a pair of means indicates a significant difference between local and regional diversity (determined via linear mixed models; Tables S9-S12).
Fig. 1
44x12mm (300 x 300 DPI)
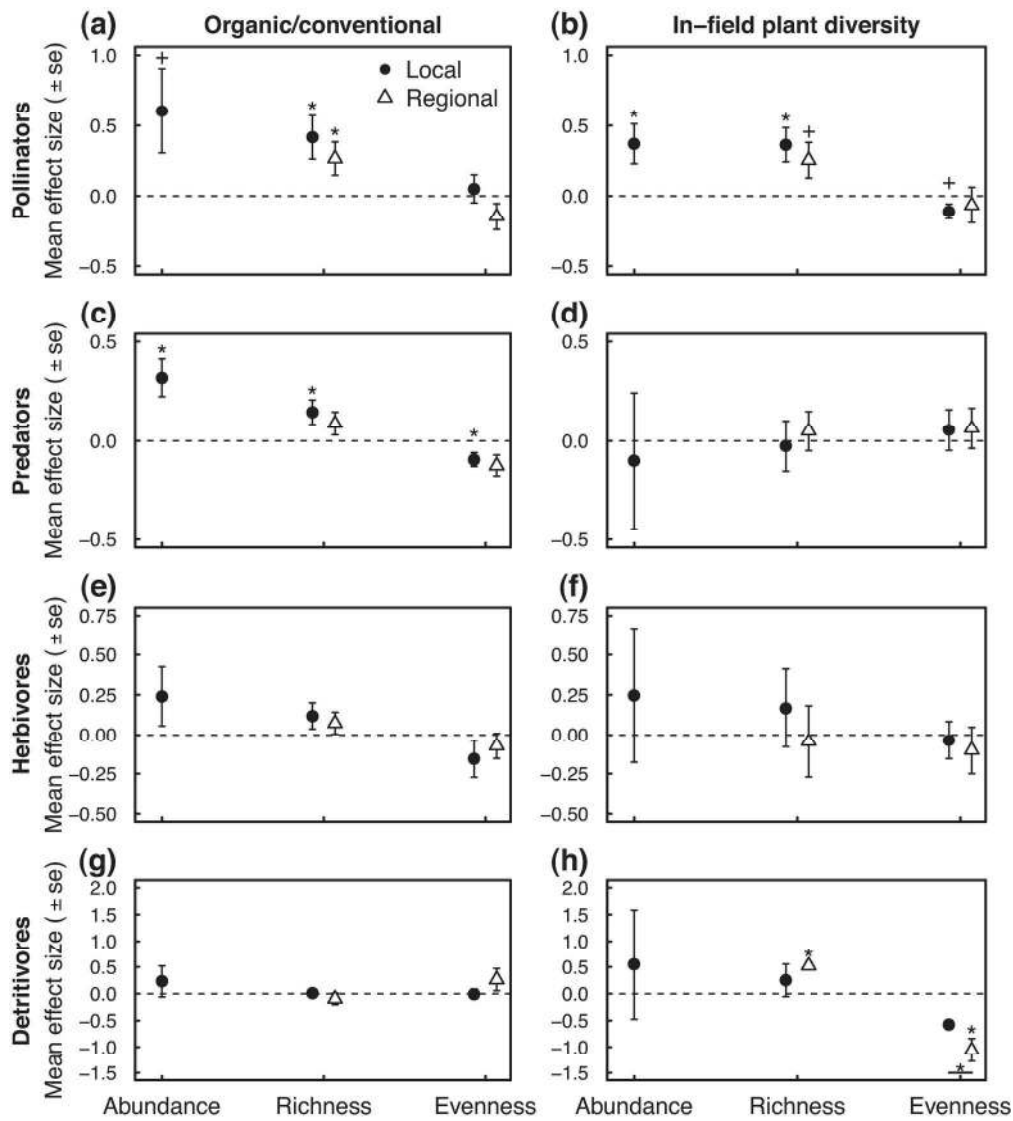
Effects of farm management schemes on abundance, local diversity, and regional diversity of arthropod functional groups. Mean log-response ratios (± SE) of (left column) adopting organic farming and (right column) promoting in-field plant diversity for (a-b) pollinators, (c-d) predators, (e-f) herbivores, and (g-h) detritivores. A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$) above a mean denotes a significant difference from zero (determined via one-sample t-tests; Tables S15-S18). Meta-regressions indicated that differences between local and regional values did not vary with functional group (Tables S9-S12).

Fig. 2
190x218mm (300 x 300 DPI)
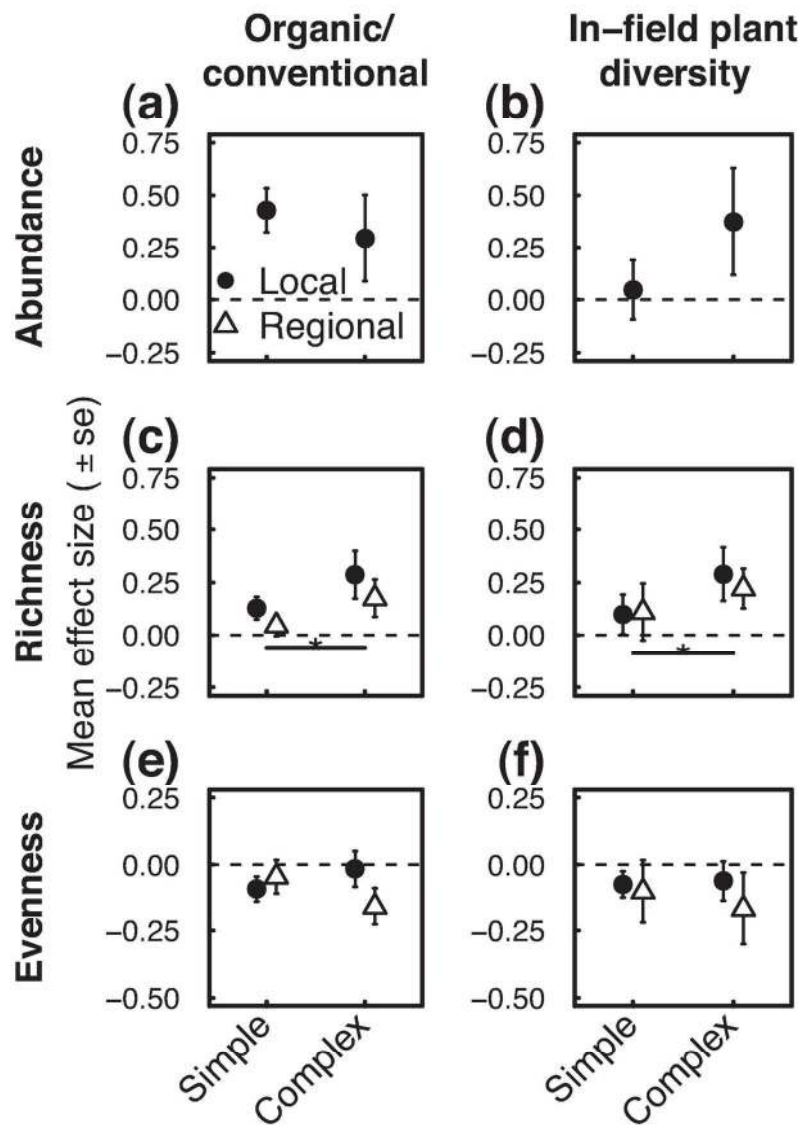
Effects of landscape complexity on the entire arthropod community in organic vs. conventional farms (left column) and fields with high vs. low in-field plant diversity (right column). Each graph shows the mean log-response ratio (± SE) for studies in simple (≤ 20% natural habitat) or complex (>20% natural habitat) landscapes for (a,b) abundance, (c,d) richness, and (e,f) evenness. A "*" (p < 0.05) or "+" (0.05 ≤ p < 0.1) below a set of means indicates a significant difference between means at the habitat complexity levels (Tables S9-S12).
Fig. 3
114x174mm (300 x 300 DPI)
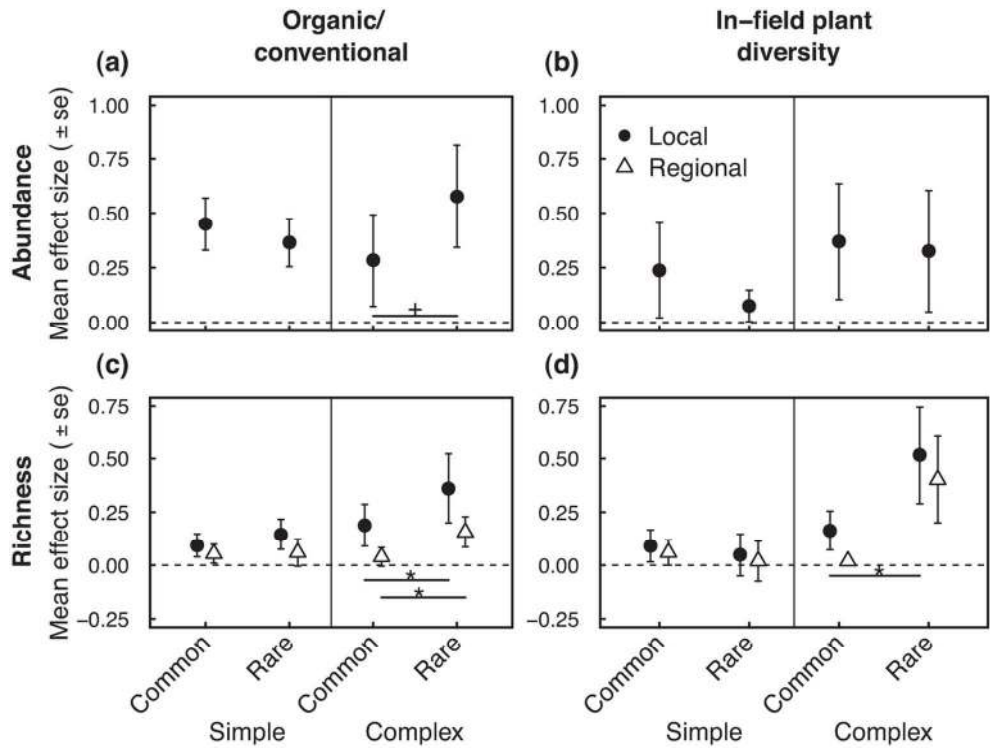
Effects of farm management schemes on abundance (a, b) and richness (c, d) of common vs. rare taxa in simple and complex landscapes. Mean log-response ratios (±SE) of (left column) adopting organic farming and (right column) promoting in-field plant diversity. A "*" ($p < 0.05$) or "+" ($0.05 \leq p < 0.1$) below a pair of means indicates a significant difference between rare and common taxa within a landscape complexity category (determined via paired t-tests; Table S19).

Fig. 4

125x93mm (300 x 300 DPI)