

1 Transferring biodiversity-ecosystem function research to the

2 management of 'real-world' ecosystems

3 Running title: Transferring BEF research

4 Peter Manning¹, Jacqueline Loos^{2,20}, Andrew D Barnes^{3,4,5}, Péter Batáry⁶, Felix J J A
5 Bianchi⁷, Nina Buchmann⁸, Gerlinde B De Deyn⁹, Anne Ebeling¹⁰, Nico Eisenhauer^{3,4},
6 Markus Fischer¹¹, Jochen Fründ¹², Ingo Grass², Johannes Isselstein¹³, Malte Jochum^{3,4,11},
7 Alexandra M Klein¹⁴, Esther O F Klingenberg¹⁵, Douglas A Landis¹⁶, Jan Lepš¹⁷, Regina
8 Lindborg¹⁸, Sebastian T Meyer¹⁹, Vicky Temperton²⁰, Catrin Westphal²¹, Teja Tschamntke².

9

- 10 1. Senckenberg Biodiversity and Climate Research Centre (BIK-F), Frankfurt am Main,
11 Germany
- 12 2. Agroecology, Department of Crop Sciences, University of Göttingen, Grisebachstr. 6,
13 37077 Göttingen, Germany
- 14 3. German Centre for Integrative Biodiversity Research (iDiv), Halle,-Jena-Leipzig,
15 Deutscher Platz 5e, 04103 Leipzig, Germany,
- 16 4. Leipzig University, Institute of Biology, Johannisallee 21-23, 04103 Leipzig
- 17 5. School of Science, University of Waikato, Private Bag 3105, Hamilton, New Zealand
- 18 6. MTA Centre for Ecological Research, Institute of Ecology and Botany Landlet
19 Landscape and Conservation Ecology Research Group, Alkotmány u. 2-4, 2163
20 Vácrátót, Hungary
- 21 7. Farming Systems Ecology, Wageningen University, Droevendaalsesteeg 1, 6708 PB
22 Wageningen, Netherlands

- 23 8. Dep. of Environmental Systems Science, ETH Zürich, Universitätstrasse 2, 8092
24 Zürich, Switzerland
- 25 9. Soil Biology Group, Wageningen University, Droevendaalsesteeg 3, 6708 PB
26 Wageningen, Netherlands
- 27 10. Institute of Ecology, Friedrich Schiller University Jena, Dornburger Str. 159, 07743
28 Jena, Germany
- 29 11. Institute of Plant Sciences, University of Bern, Altenbergrain 21, 3013 Bern,
30 Switzerland.
- 31 12. Dep. of Biometry and Environmental System Analysis, Albert-Ludwigs-University
32 Freiburg, Tennenbacher Straße 4, 79106 Freiburg. Germany
- 33 13. Institute of Grassland Science, Georg-August-University Göttingen, von-Siebold-Str.
34 8, 37075 Göttingen, Germany
- 35 14. Nature Conservation and Landscape Ecology, Albert-Ludwigs-University Freiburg,
36 Tennenbacher Straße 4, 79106 Freiburg. Germany
- 37 15. Dep. of Plant Ecology and Ecosystem Research, Georg-August University Göttingen,
38 Untere Karspüle 2, 37073 Göttingen, Germany
- 39 16. Department of Entomology and Great Lakes Bioenergy Research Center, 204 Center
40 for Integrated Plant Systems, Michigan State University , 578 Wilson Road, East
41 Lansing, Michigan 48824-1311 USA
- 42 17. Department of Botany, Faculty of Science, University of South Bohemia, Branisovska
43 31, CZ-370 05 Ceske Budejovice, Czech Republic
- 44 18. Dept. of Physical Geography, Stockholm University, 106 91 Stockholm, Sweden
- 45 19. Dep. of Ecology and Ecosystem Management, Technical University of Munich, Hans-
46 Carl-von-Carlowitz-Platz 2, 85350 Freising-Weihenstephan, Germany
- 47 20. Institute of Ecology, Faculty for Sustainability Science, Leuphana University,
48 Scharnhorststrasse 1, 21335 Lüneburg, Germany

49 21. Functional Agrobiodiversity, Department of Crop Sciences, University of Göttingen,
50 Griesebachstraße 6, 37077, Göttingen, Germany

51

52 **Abstract**

53

54 Biodiversity-ecosystem functioning (BEF) research grew rapidly following concerns that biodiversity
55 loss would negatively affect ecosystem functions and the ecosystem services they underpin. However,
56 despite evidence that biodiversity strongly affects ecosystem functioning, the influence of BEF
57 research upon policy and the management of ‘real-world’ ecosystems, i.e. semi-natural habitats and
58 agroecosystems, has been limited. Here, we address this issue by classifying BEF research into three
59 clusters based on the degree of human control over species composition and the spatial scale, in terms
60 of grain, of the study, and discussing how the research of each cluster is best suited to inform
61 particular fields of ecosystem management. Research in the first cluster, small-grain highly controlled
62 studies, is best able to provide general insights into mechanisms and to inform the management of
63 species-poor and highly managed systems such as croplands, plantations, and the restoration of
64 heavily degraded ecosystems. Research from the second cluster, small-grain observational studies, and
65 species removal and addition studies, may allow for direct predictions of the impacts of species loss in
66 specific semi-natural ecosystems. Research in the third cluster, large-grain uncontrolled studies, may
67 best inform landscape-scale management and national-scale policy. We discuss barriers to transfer
68 within each cluster and suggest how new research and knowledge exchange mechanisms may
69 overcome these challenges. To meet the potential for BEF research to address global challenges, we
70 recommend transdisciplinary research that goes beyond these current clusters and considers the social-
71 ecological context of the ecosystems in which BEF knowledge is generated. This requires recognizing
72 the social and economic value of biodiversity for ecosystem services at scales, and in units, that matter
73 to land managers and policy makers.

74 **Key words**

75 BEF research; Biodiversity experiments, Ecosystem services; Grasslands; Ecosystem management,
76 Knowledge transfer

77

78 **Introduction**

79

80 Widespread concerns over the consequences of global biodiversity loss led to an explosion of
81 ecological research in the early 1990s into the relationship between biodiversity and the functioning of
82 ecosystems (hereafter BEF research) (Schulze and Mooney, 1994; Loreau et al., 2001; Hooper et al.,
83 2005; Eisenhauer et al., 2019 this issue; Hines et al. 2019 this issue). Historically, most work in this
84 field has been conducted in experimental settings, especially in grasslands, where extinction is
85 simulated by randomly assembling plant communities differing in species and functional richness and
86 where other environmental drivers of ecosystem function are controlled for (Hector et al., 1999;
87 Tilman et al. 2001; Weisser et al. 2017). While this work has led to several robust conclusions
88 regarding the form of biodiversity-function relationships and the mechanisms that drive them
89 (Cardinale et al. 2012), there remain doubts regarding the capacity for experimental BEF research to
90 inform the management of biodiversity and ecosystem functions and services in the ‘real world’ (i.e.
91 ecosystems with communities that have not been experimentally manipulated) (Huston 1997; Lepš
92 2004, Srivistava & Vellend 2005; Wardle, 2016; Eisenhauer et al., 2016). Much of this debate
93 concerns the design of biodiversity experiments, which were established to investigate if biodiversity
94 *could* affect function, and via what mechanisms (Tilman et al., 1996; Loreau and Hector, 2001;
95 Schmid et al. 2002).

96

97 A more recent generation of BEF research has been conducted in non-experimental and naturally
98 assembled real-world ecosystems such as natural and semi-natural (hereafter semi-natural) drylands,
99 grasslands and forests (e.g., Maestre et al., 2012; Grace et al., 2016; Van Der Plas et al., 2016; Duffy
100 et al., 2017; Fanin et al., 2018; Hautier et al.; 2018, van der Plas 2019). As they are performed in

101 naturally assembled communities, shaped by both environmental drivers and global change factors,
102 these studies are correlational and tend to rely upon statistical controls, thus limiting confident
103 inference about the functional consequences of biodiversity loss in these systems. Removal
104 experiments can help overcome this issue but, to date, relatively few have been conducted (Díaz et al
105 2003; Fry et al. 2013; Fanin et al. 2018). While a lack of confident inference may limit transfer, many
106 other knowledge gaps also limit the transferability of BEF research. For example, there is little
107 consensus regarding on how strongly biodiversity loss affects ecosystem functioning, relative to other
108 drivers (Strivistava & Vellend 2005; Hooper et al 2012; Duffy et al 2017; van der Plas 2019).
109 Moreover, the functional consequences of the non-random extinction which occurs in semi-natural
110 ecosystems have largely been estimated from correlational studies (Larsen et al. 2005; Duffy et al
111 2017; van der Plas et al 2019a, but see Lyons & Schwarz 2001 and Zavaleta and Hulvey 2004).
112 Further challenges in the knowledge transfer and application of BEF research emerge from a lack of
113 information regarding the social and economic barriers to conserving biodiversity and promoting
114 diversification (Fazey et al. 2013; Rosa et al. 2019). Filling these knowledge gaps would help in
115 providing reliable evidence to inform the management of the world's ecosystems, e.g. via the
116 Intergovernmental Science-Policy Panel on Biodiversity and Ecosystem Services (IPBES) (Díaz et al.,
117 2015; Díaz et al., 2018).

118

119 In this article, we review the current understanding of the BEF relationship and discuss how BEF
120 research could inform the management of real-world ecosystems. We do this by assessing the
121 suitability of current knowledge for transfer and how this is reflected in current applied research. We
122 then identify barriers to transfer and expand on how these barriers can be overcome via future research
123 and changes to knowledge exchange mechanisms. Throughout, we emphasize the transition of BEF
124 research from a fundamental science to applied research that can inform management. By doing so we
125 assume that the promotion of certain ecosystem services is desired (e.g. carbon storage or crop
126 production).

127

128 To aid understanding of the potential transfer of BEF research, we classify it into three clusters based
129 upon a) the degree of human control over the plant community, which in experiments manifests
130 through removal of non-target species, and in real world ecosystems through management inputs, and
131 b) the size of the study plots or area, i.e. grain (Fig. 1a). While these two axes represent continuous
132 gradients, and some studies are difficult to classify, research within each cluster shares several features
133 (described below), making a general critique possible. Furthermore, each of these clusters shares
134 features with a subset of real-world ecosystems (e.g. similar levels of human control over plant
135 community and the grain of management (Fig. 1b). Based on these similarities, we suggest
136 possibilities and challenges for knowledge transfer and applications. We then identify future research
137 needs (summarized in Table 1). Throughout our discussion, we focus on terrestrial ecosystems,
138 particularly the role of plant diversity in grasslands and that of insects in agricultural landscapes. This
139 focus is a result of our own expertise and the historical focus of much BEF research on these systems
140 (Hines et al. 2019 this issue).

141 *>Figure 1 here*

142 **Small-grain and highly-controlled experiments (Cluster A)**

143

144 Since the mid 90's, more than 600 experiments have been established to explore the causal
145 relationship between biodiversity and ecosystem functioning (Cardinale et al. 2012), typically under
146 field conditions (e.g. Tilman 1996; Hector et al., 1999; Roscher et al., 2004). The primary goal of
147 these experiments was to establish whether biodiversity could affect ecosystem functioning, and so
148 they controlled for potentially confounding effects of environmental conditions, functional
149 composition, individual density, and non-random assembly and disassembly processes (Schmid et al.
150 2002, Schmid and Hector 2004, Eisenhauer et al. 2019, this issue). To achieve this, BEF experiments
151 apply a diversity treatment, where varying levels of plants species richness are sown or planted, and
152 ecosystem functioning is measured (Schmid et al. 2002: Bruhlheide et al. 2014). As such studies are
153 highly controlled (e.g. via randomized blocking, weeding and the homogenization of growing

154 conditions), diversity effects may be ascribed with confidence and detailed inferences can be made
155 regarding the identity of the mechanisms driving biodiversity effects (Loreau and Hector 2001).

156

157 While these experiments act as model systems for BEF research, with generally applicable results to a
158 wide range of systems (Schmid & Hector 2004; Eisenhauer et al., 2016), the direct application of these
159 insights in the management of real-world ecosystems could be limited for several reasons. First, the
160 sown or planted community (and its species richness) is maintained through the repeated removal of
161 non-target species, which typically does not occur in real-world systems. As a result, communities
162 may be present that would not persist without human intervention. Second, the species richness
163 gradient tends to span levels of diversity (typically 1- <20 plant species) that are much lower than
164 many semi-natural communities (Wilson et al. 2012). Third, the studies tend to be conducted in
165 replicated plots smaller than 500 m² (Tilman 1996; Hector et al., 1999; Roscher et al., 2004), with a
166 median size of 3 m² (Cardinale et al. 2012). As such studies are labor-intensive, they also tend to be
167 unreplicated at the landscape scale (but see Hector et al 1999; Kirwan et al. 2007). However, the large
168 number of experiments with comparable designs allows meta-level, large extent analyses to be
169 conducted (Balvanera et al 2006; Isbell et al. 2015; Lefcheck et al., 2015; Verheyen et al 2016; Craven
170 et al. 2018).

171

172 *What can be transferred*

173

174 BEF experiments were designed to provide general mechanistic insights into the BEF relationship.
175 Nevertheless, the close control of plant community composition and their low species diversity means
176 that findings from BEF experiments are potentially transferable to highly managed ecosystems, e.g.
177 intensive agricultural grasslands, plantation forestry, gardens, sown communities found in urban green
178 spaces or ecosystems restored from a heavily degraded state (Fig. 1b). Such systems tend to be
179 managed intensively and at small scales, e.g. via the application of selective herbicides, weeding and
180 fertilization. As these systems typically contain fewer species than most semi-natural ecosystems, we
181 predict that BEF research is best able to inform work related to diversification, rather than the impacts

182 of species loss. BEF experiment results suggest that diversification of such systems would lead to
183 considerable gains in the supply of some ecosystem services, as numerous functions related to
184 agricultural production and sustainability often increase with species diversity, including plant
185 productivity, pollination, soil carbon storage and weed suppression (Isbell et al., 2017). Moreover,
186 species-rich communities produce a more stable and constant yield (Isbell et al. 2015; Craven et al.
187 2018), which may reduce risks to farmers (Finger & Buchmann 2015).

188

189 Experimental results indicate that the benefits of diversification are greater when increasing diversity
190 from low to intermediate levels (e.g. from 1 to 8 grassland species per m²) than from medium to high
191 (e.g. from 8 to 16), as the diversity-function relationship tends to saturate (Isbell et al. 2017). As
192 species are typically grown in monocultures and in a wide range of low-diversity mixtures, data from
193 these experiments can help to identify high performing species, but also high performing mixtures, for
194 a range of ecosystem functions. Agronomists have conducted significant research on crop
195 diversification for many years (Vandermeer 1992; Brooker et al. 2015), and demonstrated that crop
196 diversification can lead to various positive outcomes, such as increased primary crop yield and
197 biocontrol (Iverson et al 2014). Moreover, intercropping can improve yield stability (Raseduzzaman &
198 Jensen 2017), and more diverse mixtures of cover crops, especially those containing legumes, lead to
199 multiple additional benefits (Storkey et al. 2015; Blesh, 2018), thus increasing their multifunctionality
200 (defined here as ecosystem service multifunctionality, the co-supply of multiple ecosystem services
201 relative to their human demand, Manning et al 2018). Similarly, crop mixtures of multiple cultivars
202 provide higher yields (Reiss and Drinkwater, 2018), and the mixing of rice varieties within a field
203 reduces disease prevalence (Zhu et al., 2000). The frameworks and fundamental insights of BEF
204 research may inform such research by identifying general rules governing complementary
205 combinations of species and varieties (Brooker et al. 2015; Wright et al. 2017).

206

207 An additional benefit of BEF experiments is that they often provide information on a wider range of
208 ecosystem services than many agricultural experiments and agronomic analyses, which tend to focus
209 on yield and its sustainability, e.g. weed control and nutrient cycling (Meyer et al. 2018). Mixtures that

210 promote the supply of multiple ecosystem services simultaneously may therefore be identified from
211 BEF studies (Storkey et al. 2015; Baeten et al. 2019). Further evidence of existing BEF transfer comes
212 from grassland studies, which indicate that there are multiple benefits of diversifying agroecosystems
213 in terms of grass yield and reduced weed abundance (Finn et al., 2013). Studies have also shown that
214 diverse grassland mixtures produce greater bioenergy yields (Khalsa et al. 2004; Tilman et al 2006).
215 However, another study of bioenergy production in grass mixtures showed that diverse mixtures were
216 not more productive than currently used monocultures, thus showing that diversification might not
217 always promote bioenergy production (Dickson and Gross, 2015). Even in the absence of positive
218 impacts of diversity on productivity, other benefits may be realized; diverse bioenergy landscapes can
219 promote the supply of other ecosystem services including greenhouse gas mitigation, pest suppression,
220 pollination, and bird watching potential (Werling et al 2014).

221

222 A number of other avenues of experimental BEF research have the capacity to inform the management
223 of intensive systems. BEF experiments show that damage to plant growth and productivity from plant
224 pathogens and pests is often weaker in more diverse communities, both aboveground (Otway et al.,
225 2005; Civitello et al., 2015) and belowground (Maron et al., 2011; Schnitzer et al., 2011).
226 Accordingly, information from BEF experiments on plant-soil feedbacks (e.g. Vogel et al. 2019a this
227 issue) could potentially help to devise effective crop rotation sequences , e.g. by identifying consistent
228 antagonistic or synergistic feedbacks between functional groups when grown together or in sequence
229 (Barel et al. 2018; Ingerslew 2018). The insights of BEF experiments are also applicable to the
230 gardens and green roof planting (Lundholm et al. 2010) and the restoration of highly degraded
231 ecosystems. Here it may be possible to determine species mixtures or particular functional trait
232 combinations, which, when sown or planted, deliver desired functions, such as soil aggregate stability
233 and soil organic matter accumulation (Lange et al. 2015; Gould et al. 2016; Kollmann et al. 2016;
234 Yang et al. 2019). In restoration, another promising approach would be to identify and sow mixtures
235 of species that facilitate each other as this is a key mechanism underlying biodiversity effects in harsh
236 environments (Wright et al., 2017). Finally, evidence from forests suggests that similar or higher
237 amounts of timber production can be achieved in mixed plantations of native species compared to

238 monocultures of plantation species, and that co-benefits, e.g. to biodiversity conservation, would also
239 be realized (Pretzsch & Schütze 2009; Hulvey et al. 2013; Gamfeldt et al. 2013; Huang et al. 2018).
240 As with crops, the results of BEF studies can also be used to indicate the tree species mixtures that
241 best achieve this multifunctionality (Teuscher et al. 2016; Baeten et al. 2019).

242

243 *Barriers to transfer and directions for future research*

244

245 While the plant communities of BEF experiments and human-dominated ecosystems share
246 similarities, there are also marked differences. For instance, the species composition in BEF
247 experiments is randomly assembled and they are usually performed in unfertilized, pesticide-free,
248 unirrigated systems. In contrast, in intensively managed real-world systems, prior knowledge has led
249 managers to select high performing, but often low diversity, mixtures by sowing and planting species
250 that deliver high levels of desired services, and/or encouraging these via pesticide application,
251 irrigation and fertilization. The benefits of diversification therefore need to be demonstrated relative to
252 these intensive low diversity communities, rather than the random low diversity assemblages found in
253 BEF experiments. For example, in European grasslands farmers typically sow or maintain mixtures of
254 a single grass, *Lolium perenne*, and a single legume, *Trifolium repens*, to which fertilizers are also
255 applied (Peeters et al. 2014). Such a mixture clearly differs from the random species-poor mixtures of
256 grassland biodiversity experiments. It is unclear if the relatively diverse and high-functioning
257 communities of biodiversity experiments are generally able to deliver yield of a similar or higher
258 quality, quantity and reliability. However, it has been demonstrated that diversification from 1-2 to 3-4
259 species provides significant increases in grassland yield and higher resistance to weed invasion
260 (Kirwan et al. 2007; Nyfeler et al. 2009; Finn et al. 2013). We hypothesize that the species-poor
261 communities found in intensively managed systems are more likely to resemble the high performing
262 species-poor communities of BEF experiments (e.g. those dominated by tall grasses of fertile
263 conditions) than the low performing communities, which may struggle to persist without regular
264 weeding and close control (e.g. those containing only a few small herbs). In contrast, the low diversity
265 situations found in experiments, where potentially dominant species are missing, could be relevant to

266 isolated habitat patches, where species cannot disperse to potentially suitable conditions and the
267 species pool is restricted.

268

269 As described above, current research suggests that links between BEF and agronomic research are
270 beginning to emerge. However, current studies do not cover the wide range of situations in which
271 diversification could be beneficial to agroecosystems. To the best of our knowledge, little work has yet
272 made the transition to widespread adoption, an exception being the standard mixtures for forage
273 production in Switzerland (see Fig. 2 for details). This lack of adoption highlights knowledge
274 exchange as an important bottleneck and another future need. To enable this, future BEF experiments
275 could increase their relevance for management by drawing experimental communities from species
276 pools that contain potentially useful and manageable species, and performing experiments in settings
277 that are similar to those found in land use systems (e.g. fertilized or grazed grasslands). In this way,
278 communities that are manageable and multifunctional may also be identified, and specific mixtures
279 can be recommended (e.g. current policy in Switzerland). These should be cost-efficient and self-
280 supporting and thus easily adapted and maintained by land managers.

281

282 Results on the relationship between biodiversity and the stability of ecosystem functions and services
283 also require re-interpretation if they are to inform ecosystem management. While definitions of
284 stability vary greatly (Wissel & Grimm 1997), BEF studies typically measure stability as the
285 coefficient of variation (e.g. Craven et al. 2018; Knapp and van der Heijden 2018), the resistance to
286 perturbations, or the rate of recovery following these (Isbell et al. 2015). In contrast, ecosystem
287 managers often perceive stability differently (Dongahue et al., 2016); while reliability is appreciated,
288 and there are minimum levels of ecosystem service supply that are acceptable and over-performance
289 (e.g. high productivity in favorable weather years, Wright et al. 2015) is often appreciated. Therefore,
290 alternative measures of stability, e.g. that measure the number of years in which the supply of services
291 exceed an acceptable threshold (Oliver et al., 2015), need to be employed if diversity-stability
292 relationships are to be determined meaningfully for agroecosystems.

293

294 Finally, the transfer of BEF research findings to the real world may be limited by the uncertainties
295 related to the profitability and management associated with diversifying species-poor communities and
296 maintaining high species richness. For example, in many agricultural grasslands, plant species loss and
297 dominance by a few nitrophilous species has occurred due to fertilization (Gaujour et al. 2012;
298 Gossner et al., 2016). Reducing nutrient availability and reversing these biodiversity declines can be
299 difficult (Smith et al., 2008; Clark and Tilman, 2010; Storkey et al., 2015). Moreover, species-rich
300 seed mixtures may prove expensive to create, and it remains to be seen if diverse and high functioning
301 grasslands can be created and maintained cost-effectively over large areas. In croplands, multispecies
302 mixtures might pose challenges to harvesting and sorting, as most modern agricultural machinery
303 specializes in managing and cropping monocultures, and the harvesting of mixtures is relatively costly
304 and labor-intensive (Magrini et al., 2011). We therefore need to know if, and under which conditions,
305 encouraging diversity in agricultural systems is efficient and feasible, especially compared to
306 management practices that deliver similar benefits (e.g. the promotion of productivity via
307 diversification versus fertilization) (Kleijn et al. 2019). A key part of this may be to acknowledge
308 additional benefits of diversity (e.g. pest control, pollination or higher yield stability) and to factor this
309 multifunctionality into comparisons. To better inform the management of agroecosystems and
310 potentially lead to their diversification, a new generation of more applied and social-ecological BEF
311 research is required (Geertsema et al., 2016). In this new work, comparisons should be made between
312 the ‘high performing low-diversity systems’ that are the current norm and multifunctional ‘sustainable
313 high-diversity systems’ that can be established and maintained at an equivalent cost to current
314 systems, or which provide additional benefits that justify greater cost (e.g. carbon storage or avoided
315 emissions) (Binder et al., 2018). Alternatively, evidence that high diversity systems can be intensified
316 without negative environmental impacts, e.g. as demonstrated for biofuel grasslands (Yang et al.
317 2018). Clearly, such approaches require transdisciplinary research involving economic and/or multiple
318 stakeholder-based assessments of the value of the diverse systems relative to current and future
319 systems and practices (Jackson et al. 2012; Geertsema et al., 2016; Bretagnolle et al. 2018; Kleijn et
320 al. 2019) (Table 1).

321 **Small-grain studies with low experimental control (Cluster B)**

322

323 The second cluster contains small-grain observational studies that investigate natural- or human-
324 induced gradients of plant diversity in less intensively managed systems (e.g. Kahmen et al. 2005a;
325 Maestre et al., 2012; Soliveres et al., 2016a; van der Plas et al., 2016; Zhu et al., 2016) (Fig. 1). In this
326 cluster, we also consider experiments in which particular species or functional groups are removed
327 from intact ecosystems, often according to simulated global change scenarios (Smith & Knapp 2003;
328 Cross & Harte, 2007; Suding et al., 2008, Fry et al. 2013; Pan et al. 2016; Fanin et al. 2018), and those
329 which boost diversity in established communities or disturbed sites, e.g. via seeding (van der Putten et
330 al. 2000; Bullock et al 2007; Stein et al. 2008; Weidlich et al. 2018). Finally, we also consider global
331 change driver experiments, where biodiversity change is treated as a co-variate and used to explain
332 observed changes in function (e.g. Grace et al., 2016; Hautier et al., 2018). Plot sizes are similar to
333 those in cluster A (i.e. $<500\text{m}^2$) and diversity levels vary greatly, from inherently species-poor
334 ecosystems (e.g. Suding et al., 2008) to species-rich communities (Allan et al., 2015). Therefore, in
335 contrast to most of the experiments of cluster A, studies from cluster B tend to contain more mature
336 communities with higher species richness, fewer monocultures, less or no weeding, and species
337 compositions and management regimes that are more similar to real-world low management intensity
338 systems. In most of these studies, and in contrast to most BEF experiments that manipulate random
339 community assembly, diversity loss occurs as non-random disassembly in response to environmental
340 drivers. Observational studies of cluster B often statistically control for co-varying factors that may
341 also drive ecosystem functions. These may include biotic covariates, such as functional composition
342 and the abundance of different functional groups (Maestre et al., 2012; Allan et al., 2015; Soliveres et
343 al., 2016a; Soliveres et al., 2016b; Van Der Plas et al., 2016), which strongly co-vary with diversity in
344 many communities (Allan et al., 2015; Barnes et al. 2016; Soliveres et al., 2016).

345

346 The design of studies in this cluster limits interpretation about the cause of biodiversity effects as data
347 for monoculture performances are usually unavailable, meaning that the mechanisms underlying
348 biodiversity effects cannot be estimated (Loreau and Hector 2001). This is unfortunate as these

349 processes may differ in their strength compared to biodiversity experiments. For example, in mature
350 communities, species may show higher levels of niche differentiation at both between and within
351 species levels (Zuppinge-Dingley et al., 2014; Guimarães-Steinicke et al. 2019, this issue). A final
352 property differentiating cluster B studies from those of cluster A is that variation in the diversity of
353 other trophic levels is a complex product of responses to environmental drivers and concurrent
354 changes in all trophic levels (Tschardt et al., 2005; Soliveres et al. 2016a,b), rather than primarily
355 driven by variation in the diversity of primary producers (Scherber et al. 2010).

356

357 *What can be transferred*

358

359 Because they are conducted in unmanipulated real-world ecosystems, cluster B results are directly
360 transferable to semi-natural ecosystems, which experience species loss and compositional change due
361 to global environmental change. Cluster B studies provide direct estimates of the real-world impacts of
362 global change drivers on diversity, and the corresponding impact of these changes on ecosystem
363 function. However, most cluster B studies are observational, so patterns remain correlational, despite
364 statistical controls. Nevertheless, due to their greater realism, syntheses of cluster B results (van der
365 Plas 2019a), can provide statistical estimates of where different components of biodiversity play their
366 greatest role, and estimates may be used as an evidence base for both local managers and in global
367 assessments.

368

369 The experimental studies of cluster B can provide information on how diversification can boost
370 ecosystem functioning in restored or enriched communities. For example, several studies show that
371 sowing into intact communities can increase both species richness and ecosystem functioning,
372 including community productivity and carbon storage (Bullock et al 2007; Stein et al. 2008; Weidlich
373 et al. 2018).

374

375 *Barriers to transfer and directions for future research*

376

377 For research in cluster B to become more directly transferable to the management of semi-natural
378 ecosystems, greater confidence in the mechanisms underlying real-world BEF relationships is needed.
379 While management recommendations may be drawn from selected case studies such as those
380 presented above, a general understanding of the relative and interacting roles of environmental
381 covariates, direct effects of global change drivers and various facets of diversity and compositional
382 change is lacking (van der Plas 2019a). Biodiversity could play an important role in maintaining
383 ecosystem function in real world ecosystems. Yet, whether loss of a few species at this scale makes a
384 strong contribution to function, relative to these other drivers, has been only been tested in a limited
385 number of cases (e.g. Manning et al. 2006; Allan et al. 2015; Winfree et al. 2015; Grace et al., 2016),
386 and inconsistently, making generalization difficult (van der Plas 2019a). To address this issue,
387 observational studies need to ensure that factors such as abundance and functional composition are
388 properly controlled for statistically. Predictions of the impacts of drivers on ecosystem services can be
389 made by combining a) estimates of expected biodiversity change according to different global change
390 drivers across a range of conditions (e.g. Grace et al., 2016; Hautier et al., 2018; Bjorkman et al.
391 2018), b) knowledge of how great a difference to functions and services such changes will make (e.g.
392 Craven et al. 2018), and c) ecosystem service production functions (Isbell et al. 2015). This in turn
393 allows for estimates of where ecosystem service-based arguments for conservation are strongest. Such
394 predictions, if verified, could then form a sound basis for management decisions.

395

396 Transfer would also be enabled by a new generation of experiments. These could include a wider
397 range of non-random extinction scenarios, assessments of the relative importance of abiotic drivers of
398 function and biodiversity (e.g. Manning et al., 2006; Isbell et al., 2013), and the reduction of diversity
399 from high to intermediate levels (Zobel et al. 1994), in order to verify, or refute the results of
400 observational studies. To do this, manipulations such as the manipulation of dominance and functional
401 composition, trait dissimilarity, or other aspects of biodiversity could be employed (Smith and Knapp,
402 2003; Manning et al., 2006; Cross and Harte, 2007). Manipulations that simulate the homogenization
403 of biota (i.e. the loss of beta diversity, while alpha diversity remains unchanged), may also prove
404 informative, as this may be as, or more, common than alpha diversity loss in real-world ecosystems

405 (Flohre et al., 2011; Vellend et al., 2014; Dornelas et al., 2014; Gossner et al., 2016; Wardle 2016).
406 Finally, it may be possible to link community assembly mechanisms (e.g. founder effects and habitat
407 filtering) and functional BEF research to identify how to increase species richness and promote certain
408 ecosystem functions, information that would be particularly useful in ecosystem restoration (Bullock
409 et al. 2007; Stein et al. 2008; Kirmer et al. 2012; Weidlich et al. 2018) (Table 1).

410

411 Work is also needed in converting the measures of ecosystem function commonly taken in ecological
412 studies into measures of ecosystem services that are of relevance to stakeholders (Mace et al. 2012;
413 Kleijn et al. 2019). This requires the development of new metrics, e.g. trait measures that link to
414 nutritional quality or cultural services such as aesthetic appeal. Applied studies could explicitly
415 measure relevant ecosystem services, e.g. by involving stakeholders, assessing which services are
416 most important to them, and adapting function measures to quantify these (Martín-López et al. 2012;
417 King et al. 2015; Manning et al. 2018). This approach, and many of the others outlined above requires
418 inter- and transdisciplinary research involving stakeholders and researchers from other disciplines e.g.
419 with farmers, local governments, agronomists and economists.

420 **Large-grain studies without experimental control (Cluster C)**

421

422 The third cluster (C) contains BEF studies that cover large areas (from 100 m² to landscapes) (e.g.
423 Larsen et al. 2005; Garibaldi et al. 2013; Winfree et al., 2018). Due to the huge efforts required to
424 manipulate diversity at a large spatial and temporal grain (Teuscher et al., 2016), such studies tend to
425 be observational, comparative, and of low replication, although the large number of such studies has
426 allowed for meta-level analyses to be conducted (Lichtenberg et al. 2017). The focal study organisms
427 also tend to be invertebrates, particularly pollinators, instead of plants. The measurement of
428 biodiversity (e.g. species richness and functional diversity) is also often limited in these studies due to
429 the effort required to measure it directly over large areas. As a result, it is often landscape variables,
430 such as landscape configuration and the proportion of different land uses that are related to function,
431 rather than diversity (e.g. Boserup et al. 2017; Hass et al., 2018). These landscape properties

432 may influence the dispersal, abundance and diversity of organisms within the landscape, and may also
433 correlate with management factors and abiotic drivers of ecosystem function (Gómez-Virués et al.,
434 2015; Dominik et al., 2018; Lindborg et al., 2017). As a result of these covariances, the role of
435 biodiversity in driving ecosystem functioning cannot always be confidently ascribed (Tscharntke et al
436 2016).

437

438 Within this cluster, we also place remote sensing studies (e.g. Oehri et al., 2017) and national and
439 regional correlational studies (e.g. Anderson et al., 2009). In these, biodiversity can only be measured
440 using proxies or with presence/absence data within large grid cells (e.g. 10 x 10 km), e.g. from
441 national monitoring schemes. These coarse biodiversity measures are then correlated with ecosystem
442 service proxy measures such as carbon storage and recreational use. These studies often lack a strong
443 mechanistic basis, and focus instead on how biodiversity co-varies with ecosystem services (e.g.
444 Anderson et al., 2009; Maskell et al., 2013). Even where covariates are included and mechanistic
445 relationships postulated (e.g. Oehri et al., 2017; Duffy et al., 2017), causal links are hard to infer due
446 to the strong covariance between biodiversity and other drivers, and the high probability of missing, or
447 improperly measuring, important covariates.

448

449 Another common type of BEF study at this scale are those showing that functional biodiversity co-
450 varies or differs across environmental gradients and management regimes (Rader et al., 2014, Gómez-
451 Virués et al., 2015). While there is significant evidence that functional traits do relate to ecosystem
452 processes and properties at landscape and national scales (e.g. Lavorel et al. 2011; Garibaldi et al.
453 2015; Manning et al. 2015), evidence for a mechanistic link between the functional diversity of traits
454 to the supply of ecosystem services at these scales is generally limited.

455

456 *What can be transferred*

457

458 As the studies of cluster C are performed in real landscapes, and as management is often conducted at
459 large scales (e.g. by farmers or foresters), research findings from this cluster are potentially of high

460 relevance to policy and large-scale management, e.g. via payments for ecosystem service schemes. In
461 recent years, a number of studies have demonstrated large-scale benefits of landscapes with high
462 diversity of crops and non-crop habitats, which support higher biodiversity (Gardiner et al., 2009;
463 Redlich et al., 2018). These benefits include more effective pollination and biological pest control
464 (Garibaldi et al. 2013; Winfree et al., 2018). By showing how diversity and diversification practices
465 influence ecosystem service delivery, these practices can then be incorporated into agronomic
466 considerations (Rosa et al., 2019) and into agri-environment policy (Garibaldi et al. 2014). Studies at
467 this scale also complement those of the other clusters by showing that biodiversity not only promotes
468 ecosystem function and services at the plot scale but also via spillover effects into the surrounding
469 landscape, with ecosystem service benefits including pest suppression, pollination, and bird watching
470 potential (Blitzer et al 2012; Werling et al 2014). However, biodiversity does not always promote
471 function at these scales. For example, natural enemy diversity does not always relate to pest
472 abundance, nor higher crop yields (Tschardt et al. 2016), and in some cases biodiversity does not
473 control pests as effectively as pesticides (Samneggard et al. 2018).

474

475 *Barriers to transfer and directions for future research*

476

477 The observational nature of most research in this cluster means that the exact role of diversity in
478 driving ecosystem function and providing ecosystem services at these scales is hard to ascertain. This
479 general limitation is compounded by several other barriers which can prevent transfer to landscape
480 management and policy. First, several processes could drive BEF relationships at landscape scales that
481 do not operate at the smaller grain size of clusters A and B, and as a result are little acknowledged in
482 BEF research, outside of theory (Loreau et al., 2003; Tschardt et al., 2012; Lindborg et al., 2017).
483 These include the spatial processes that maintain diversity, the matching between species and
484 environmental conditions in which they perform well (Leibold et al. 2017; Mori et al 2018), and the
485 potential for different species to provide different functions and services in different patches of the
486 landscape, thus boosting landscape multifunctionality (van der Plas et al. 2016, 2019b). The strength
487 and role of such mechanisms clearly needs to be demonstrated. Another key problem in transferring

488 BEF research to large scales is that landscape managers typically seek to simultaneously promote
489 multiple ecosystem services, i.e. the multifunctionality of landscapes, not single ecosystem functions
490 at the plot scale (Manning et al., 2018; Kremen & Merenlender 2018). A focus on single functions is
491 problematic if they trade-off and the components of diversity that boost some ecosystem services
492 diminish others. For example, the maintenance of biodiversity-rich habitats may add resilience to
493 multiple ecosystem functions at the landscape scale, but also occupies land that could be used for crop
494 production.

495

496 New research approaches are required to overcome the difficulties in identifying how biodiversity
497 controls ecosystem functioning at large scales, and how biodiversity may be conserved and promoted
498 to increase the supply of ecosystem services. First, to ensure that service measures are of relevance to
499 stakeholders, we require a better understanding of which services are demanded by different
500 stakeholders, and at which different temporal and spatial scales, so that relevant indicator variables or
501 ecosystem service production functions can be used (Tallis 2011). A more holistic approach, which
502 accounts for the relative demand for different ecosystem services and how this changes with socio-
503 economic context, is therefore required, e.g. to assess how much land can be returned to a high
504 biodiversity condition while maintaining desired levels of food production and other ecosystem
505 services (Clough et al. 2011; Kremen & Merenlender 2018; Manning et al. 2018). Such studies should
506 also identify what drives patterns of land use and management and hence biodiversity loss, so that
507 appropriate interventions can be identified (Grass et al. 2019).

508

509 To consider landscape multifunctionality and its dependence on biodiversity, multiple ecosystem
510 services need to be scaled up in space and time, which is challenging. Some of the functions that can
511 be measured at the plot scale can be ‘linearly’ scaled up, e.g. by using remote sensing proxies of
512 diversity and functional traits, and interpolated maps, e.g. of climate and soil properties (Manning et
513 al., 2015; van der Plas et al., 2018). Others, however, require an understanding of spatial interactions
514 that makes their upscaling more complex, e.g. pollination and nutrient leaching (Koh et al. 2016,
515 Lindborg et al. 2017.). Furthermore, some services that operate at large scales (e.g. flood control,

516 landscape aesthetics) cannot be predicted and scaled up from small-scale measures. Therefore, new
517 procedures and methods are needed to quantify large-scale multifunctionality and the role of
518 biodiversity in driving it. There have been calls for landscape-scale experiments to address these
519 issues (Koh et al., 2009; Landis 2017). One example is the recent EFForTS project in which "tree
520 islands" of varying size and tree diversity (0-6 species) have been planted in oil-palm clearings
521 (Teuscher et al., 2016). Initial results indicate no economic trade-off: the islands generate yield gains
522 which compensate for the reduced number of oil palms (Gerard et al., 2017). However, the high
523 financial cost and/or logistical effort of such experiments means it may be more realistic to use
524 biophysical models in most cases. Unfortunately, such models do not currently fully represent the
525 complexity of biodiversity or its relationship with ecosystem functions and services (Lavorel et al.
526 2017).

527

528 To understand biodiversity-landscape multifunctionality relationships, a greater knowledge of which
529 aspects of diversity underpin different ecosystem services is also required. While knowledge exists
530 regarding the drivers of many ecosystem service provider groups at the landscape scale (e.g. plants,
531 birds, butterflies and pollinators, Roschewitz et al. 2005; Rösch et al. 2015; Kormann et al 2015; Grab
532 et al. 2019), this understanding needs to be extended to other groups, including soil microbes and soil
533 fauna. Similarly, understanding of how spatial biodiversity dynamics affect functions and the services
534 they underpin needs to be extended to taxa involved in services other than pest control and pollination
535 (Table 1). In some cases, there may be trade-offs between services, e.g. if the conditions that
536 maximize the diversity of one taxa do not favor another (van der Plas 2019b). This research may also
537 demonstrate that when it comes to real-world ecosystem services and landscape-level
538 multifunctionality, biodiversity effects are not easily generalizable, but depend on the context. Thus,
539 the rules of this context-dependency need to be identified (Allan et al 2015; Birkhofer et al., 2018;
540 Samnegard et al 2018). Doing this will limit uncertainty; managers could be less reluctant to manage
541 for biodiversity when the degree to which it provides ecosystem service benefits at larger scales has
542 been clearly demonstrated. In semi-natural ecosystems the promotion of the biodiversity components
543 underpinning ecosystem services are most likely to be achieved via management options that are

544 simple and effective over large areas, and so the practices that would promote the desired facets of
545 biodiversity, e.g. mowing or the introduction of selective grazers, may need to be identified.

546 **Conclusion**

547

548 A vast array of BEF studies has taught us much about the complex relationship between biodiversity
549 and ecosystem functioning. In this article, we argue that with some re-analysis and re-interpretation,
550 some of this research could be directly transferred to policy and management, where practitioners
551 could use its insights to guide the diversification of agricultural and other human-dominated
552 ecosystems, and inform the conservation of biodiversity in semi-natural ecosystems. However, there
553 are numerous challenges to the transfer of BEF research to more applied research and practice, and we
554 argue that these challenges differ depending on the spatial grain of the study and the degree of
555 community manipulation. While acknowledging the differences in transferability between these
556 clusters of BEF research may help resolve the ongoing debate about relevance of BEF findings a new
557 generation of BEF research is also required. This would involve the merging and connecting research
558 between the current clusters, e.g. the setup of a new generation of biodiversity experiments that bridge
559 the gap between current BEF experiments and observational studies. These should be complemented
560 by new observational studies which more comprehensively account for covarying factors and which
561 better acknowledge the link between ecosystem function and ecosystem services (Table 1).

562

563 It should be noted that the main message transferred from BEF research may simply be a stronger and
564 more confident argument that it is important to conserve the diversity that is already present in semi-
565 natural systems. In some cases BEF research may also show that not every species plays a positive or
566 strong role in driving certain ecosystem functions, and that a small number of species dominate the
567 supply of certain services (Kleijn *et al.* 2015). In such cases, acknowledging the non-market benefits
568 of species and returning to more traditional ethical arguments will help promote biodiversity
569 conservation (e.g. Hill *et al.* 2019).

570 Finally, to make BEF research more applied, large-scale studies that utilise novel approaches to
571 investigate the role of diversity in providing the desired ecosystem services at the landscape scale are
572 required (Table 1). Accordingly, key considerations in applied BEF research are to acknowledge when
573 research is fundamental or applied, and to clarify when services, rather than functions, are being
574 considered, thus making it transparent which services and functions are focal and why, and
575 acknowledging which stakeholder groups may benefit. In many respects, the technical solutions to the
576 challenges addressed in this article are already being investigated. However, if the potential for BEF
577 research to address global challenges is to be fully realized, future BEF must also be transdisciplinary,
578 and include the main stakeholders of the ecosystem collaboratively from their inception. By
579 considering social-ecological context, BEF research should be better able to demonstrate the social
580 and economic value of biodiversity at the scales that matter to land managers and policy makers.

581

582 **Acknowledgements**

583

584 This work was funded by Deutsche Forschungsgemeinschaft; DFG, German Research Foundation
585 Grant Ei 862/13 to MF, NB, AK, NE and TT. The Jena Experiment is funded by the Deutsche
586 Forschungsgemeinschaft (DFG, German Research Foundation; FOR 1451), the Friedrich Schiller
587 University Jena, the Max Planck Institute for Biogeochemistry in Jena, and the Swiss National Science
588 Foundation. NE ADB and MJ acknowledge support by the German Centre for Integrative Biodiversity
589 Research (iDiv) Halle-Jena-Leipzig (DFG FZT 118). DAL acknowledges support from Great Lakes
590 Bioenergy Research Center, U.S. Department of Energy, Office of Science, Office of Biological and
591 Environmental Research (Awards DE-SC0018409 and DE-FC02-07ER64494), by the National
592 Science Foundation Long-term Ecological Research Program (DEB 1637653) at the Kellogg
593 Biological Station, and by Michigan State University AgBioResearch. CW is grateful for funding by
594 the Deutsche Forschungsgemeinschaft (DFG) (Project number 405945293).

595

596 **References**

597

598 AGFF. 2019. <http://www.agff.ch/deutsch/aktuell.html>
599
600 Agrarforschung Schweiz 2019. https://www.agrarforschungschweiz.ch/archiv_11en.php?id_artikel=2244
601
602 Allan, E., Manning, P., Alt, F., Binkenstein, J., Blaser, S., Blüthgen, N., Böhm, S., Grassein, F., Hölzel, N.,
603 Klaus, V.H. and Kleinebecker, T., 2015. Land use intensification alters ecosystem multifunctionality via loss of
604 biodiversity and changes to functional composition. *Ecology Letters*, 18, 834-843.
605
606 Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B. and
607 Gaston, K.J., 2009. Spatial covariance between biodiversity and other ecosystem service priorities. *Journal of*
608 *Applied Ecology*, 46, 888-896.
609
610 Baeten, L., Verheyen, K., Wirth, C., Bruelheide, H., Bussotti, F., Finér, L., Jaroszewicz, B., Selvi, F., Valladares,
611 F., Allan, E., 2013. A novel comparative research platform designed to determine the functional significance of
612 tree species diversity in European forests. *Perspectives in Plant Ecology, Evolution and Systematics* 15, 281-291.
613
614 Baeten, L., Bruelheide, H., van der Plas, F., Kambach, S., Ratcliffe, S., Jucker, T., Allan, E., Ampoorter, E.,
615 Barbaro, L., Bastias, C.C. and Bauhus, J., 2019. Identifying the tree species compositions that maximize
616 ecosystem functioning in European forests. *Journal of Applied Ecology*, 56, 733-744.
617
618 Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.S., Nakashizuka, T., Raffaelli, D. and Schmid, B., 2006.
619 Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*, 9,
620 1146-1156.
621
622 Barel, J.M., Kuyper, T.W., de Boer, W., Douma, J.C. and De Deyn, G.B., 2018. Legacy effects of diversity in
623 space and time driven by winter cover crop biomass and nitrogen concentration. *Journal of Applied Ecology*, 55,
624 299-310.
625
626 Barnes, A.D., Weigelt, P., Jochum, M., Ott, D., Hodapp, D., Haneda, N.F. and Brose, U., 2016. Species richness
627 and biomass explain spatial turnover in ecosystem functioning across tropical and temperate ecosystems.
628 *Philosophical Transactions of the Royal Society B: Biological Sciences*, 371, p.20150279.

629

630 Binder, S., Isbell, F., Polasky, S., Catford, J.A. and Tilman, D., 2018. Grassland biodiversity can pay.

631 *Proceedings of the National Academy of Sciences*, 115, 3876-3881.

632

633 Birkhofer, K., Andersson, G.K., Bengtsson, J., Bommarco, R., Dänhardt, J., Ekbom, B., Ekroos, J., Hahn, T.,

634 Hedlund, K., Jönsson, A.M. and Lindborg, R., 2018. Relationships between multiple biodiversity components

635 and ecosystem services along a landscape complexity gradient. *Biological Conservation*, 218, 247-253.

636

637 Bjorkman, A.D., Myers-Smith, I.H., Elmendorf, S.C., Normand, S., Rüger, N., Beck, P.S., Blach-Overgaard, A.,

638 Blok, D., Cornelissen, J.H.C., Forbes, B.C. and Georges, D., et al. 2018. Plant functional trait change across a

639 warming tundra biome. *Nature*, 562, 57.

640

641 Blaauw, B.R. and Isaacs, R., 2014. Flower plantings increase wild bee abundance and the pollination services

642 provided to a pollination-dependent crop. *Journal of Applied Ecology*, 51, 890-898.

643

644 Blitzer, E.J., Dormann, C.F., Holzschuh, A., Klein, A.M., Rand, T.A. and Tschardtke, T., 2012. Spillover of

645 functionally important organisms between managed and natural habitats. *Agriculture, Ecosystems &*

646 *Environment*, 146, 34-43.

647

648 Blesh, J., 2018. Functional traits in cover crop mixtures: Biological nitrogen fixation and multifunctionality.

649 *Journal of Applied Ecology*, 55, 38-48.

650

651 Bosem Baillod, A., Tschardtke, T., Clough, Y., Batáry, P., 2017. Landscape-scale interactions of spatial and

652 temporal cropland heterogeneity drive biological control of cereal aphids. *Journal of Applied Ecology* 54, 1804–

653 1813.

654

655 Bretagnolle, V., Berthet, E., Gross, N., Gauffre, B., Plumejeaud, C., Houte, S., Badenhassser, I., Monceau, K.,

656 Allier, F., Monestiez, P. and Gaba, S., 2018. Towards sustainable and multifunctional agriculture in farmland

657 landscapes: lessons from the integrative approach of a French LTSER platform. *Science of the Total*

658 *Environment*, 627, 822-834.

659

660 Brooker, R.W., Bennett, A.E., Cong, W.F., Daniell, T.J., George, T.S., Hallett, P.D., Hawes, C., Iannetta, P.P.,
661 Jones, H.G., Karley, A.J. and Li, L., 2015. Improving intercropping: a synthesis of research in agronomy, plant
662 physiology and ecology. *New Phytologist*, 206, 107-117.
663
664 Bruelheide, H., Nadrowski, K., Assmann, T., Bauhus, J., Both, S., Buscot, F., Chen, X.Y., Ding, B., Durka, W.,
665 Erfmeier, A. and Gutknecht, J.L., 2014. Designing forest biodiversity experiments: general considerations
666 illustrated by a new large experiment in subtropical China. *Methods in Ecology and Evolution*, 5, 74-89.
667
668 Bullock, J.M., Pywell, R.F. and Walker, K.J., 2007. Long-term enhancement of agricultural production by
669 restoration of biodiversity. *Journal of Applied Ecology*, 44, 6-12.
670
671 Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M.,
672 Tilman, D., Wardle, D.A. and Kinzig, A.P., 2012. Biodiversity loss and its impact on humanity. *Nature*, 486,
673 p.59.
674
675 Civitello, D.J., Cohen, J., Fatima, H., Halstead, N.T., Liriano, J., McMahon, T.A., Ortega, C.N., Sauer, E.L.,
676 Sehgal, T., Young, S. and Rohr, J.R., 2015. Biodiversity inhibits parasites: broad evidence for the dilution effect.
677 *Proceedings of the National Academy of Sciences*, 112,8667-8671.
678
679 Clark, C.M. and Tilman, D., 2010. Recovery of plant diversity following N cessation: effects of recruitment,
680 litter, and elevated N cycling. *Ecology*, 91, 3620-3630.
681
682 Clec'h S, Huber R, Buchmann N, Gosal A, Hörtnagl L, Huguenin-Elie O, Jeanneret P, Lüscher A, Schneider M,
683 Finger R. Assessment of spatial variability of multiple ecosystem services in grasslands of different intensities.
684 *Journal of Environmental Management* (submitted Feb. 2019)
685
686 Clough, Y., Barkmann, J., Juhbandt, J., Kessler, M., Wanger, T.C., Anshary, A., Buchori, D., Cicuzza, D.,
687 Darras, K., Putra, D.D. and Erasmi, S., et al. 2011. Combining high biodiversity with high yields in tropical
688 agroforests. *Proceedings of the National Academy of Sciences*, 108, 8311-8316.
689

690 Clough, Y., Krishna, V.V., Corre, M.D., Darras, K., Denmead, L.H., Mejjide, A., Moser, S., Musshoff, O.,
691 Steinebach, S., Veldkamp, E. and Allen, K., et al. 2016. Land-use choices follow profitability at the expense of
692 ecological functions in Indonesian smallholder landscapes. *Nature Communications*, 7, 13137.
693
694 Conservation Evidence <https://www.conservationevidence.com/>
695
696 Craven, D., Eisenhauer, N., Pearse, W.D., Hautier, Y., Roscher, C., Isbell, F., Bahn, M., Beierkuhnlein, C.,
697 Bönisch, G., Buchmann, N. and Byun, C., et al. 2018. Multiple Facets of Biodiversity Drive the Diversity-
698 Stability Relationship. *Nature Ecology & Evolution*, 2, 1.
699
700 Cross, M.S. and Harte, J., 2007. Compensatory responses to loss of warming-sensitive plant species. *Ecology*,
701 88, 740-748.
702
703 Díaz, S., Symstad, A.J., Chapin III, F.S., Wardle, D.A. and Huenneke, L.F., 2003. Functional diversity revealed
704 by removal experiments. *Trends in Ecology & Evolution*, 18, 140-146.
705
706 Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J.R., Arico, S.,
707 Báldi, A. and Bartuska, A., et al. 2015. The IPBES Conceptual Framework—connecting nature and people.
708 *Current Opinion in Environmental Sustainability*, 14, 1-16..
709
710 Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z., Hill, R., Chan, K.M., Baste,
711 I.A., Brauman, K.A. and Polasky, S., 2018. Assessing nature's contributions to people. *Science*, 359, 270-272.
712
713 Dickson, T.L. and Gross, K.L., 2015. Can the results of biodiversity-ecosystem productivity studies be translated
714 to bioenergy production?. *PloS one*, 10, e0135253.
715
716 Dominik, C., Seppelt, R., Horgan, F.G., Settele, J. and Václavík, T., 2018. Landscape composition,
717 configuration, and trophic interactions shape arthropod communities in rice agroecosystems. *Journal of Applied*
718 *Ecology*, 55, 2461-2472.
719

720 Donohue, I. , Hillebrand, H. , Montoya, J. M., Petchey, O. L., Pimm, S. L., Fowler, M. S., Healy, K. , Jackson,
721 A. L., Lurgi, M. , McClean, D. , O'Connor, N. E., O'Gorman, E. J., Yang, Q. and Adler, F. (2016), Navigating
722 the complexity of ecological stability. *Ecology Letters*, 19, 1172-1185.

723

724 Dornelas, M., Gotelli, N.J., McGill, B., Shimadzu, H., Moyes, F., Sievers, C. and Magurran, A.E., 2014.
725 Assemblage time series reveal biodiversity change but not systematic loss. *Science*, 344, 296-299.

726

727 Duffy, J.E., Godwin, C.M. and Cardinale, B.J., 2017. Biodiversity effects in the wild are common and as strong
728 as key drivers of productivity. *Nature*, 549, 261.

729

730 Eisenhauer, N., Barnes, A.D., Cesarz, S., Craven, D., Ferlian, O., Gottschall, F., Hines, J., Sendek, A., Siebert,
731 J., Thakur, M.P. and Türke, M., 2016. Biodiversity–ecosystem function experiments reveal the mechanisms
732 underlying the consequences of biodiversity change in real world ecosystems. *Journal of Vegetation Science*, 27,
733 1061-1070.

734

735 Eisenhauer, N., Sendek, A., Siebert, J., Thakur, M.P., 2016. Biodiversity–ecosystem function experiments reveal
736 the mechanisms underlying the consequences of biodiversity change in real world ecosystems. *Journal of*
737 *Vegetation Science* 27, 1061-1070.

738

739 **Eisenhauer et al., 2019 this issue**

740

741 European Landowners Organisation (ELO) <https://www.europeanlandowners.org/>

742

743 Emmerson, M., Morales, M.B., Oñate, J.J., Batáry, P., Berendse, F., Liira, J., Aavik, T., Guerrero, I., Bommarco,
744 R., Eggers, S. and Pärt, T., 2016. How agricultural intensification affects biodiversity and ecosystem services. In
745 *Advances in Ecological Research*, 55, 43-97. Academic Press.

746

747 Flohre, A., Fischer, C., Aavik, T., Bengtsson, J., Berendse, F., Bommarco, R., Ceryngier, P., Clement, L.W.,
748 Dennis, C., Eggers, S. and Emmerson, M., 2011. Agricultural intensification and biodiversity partitioning in
749 European landscapes comparing plants, carabids, and birds. *Ecological Applications*, 21, 1772-1781.

750

751 Fanin, N., Gundale, M.J., Farrell, M., Ciobanu, M., Baldock, J.A., Nilsson, M.C., Kardol, P. and Wardle, D.A.,
752 2018. Consistent effects of biodiversity loss on multifunctionality across contrasting ecosystems. *Nature Ecology*
753 *and Evolution*, 2, 269.

754

755 Fazey, I., Evely, A.C., Reed, M.S., Stringer, L.C., Kruijssen, J., White, P.C., Newsham, A., Jin, L., Cortazzi, M.,
756 Phillipson, J. and Blackstock, K., 2013. Knowledge exchange: a review and research agenda for environmental
757 management. *Environmental Conservation*, 40, 19-36.

758

759 Finger, R. and Buchmann, N., 2015. An ecological economic assessment of risk-reducing effects of species
760 diversity in managed grasslands. *Ecological Economics*, 110, 89-97.

761

762 Finn, J.A., Kirwan, L., Connolly, J., Sebastià, M.T., Helgadottir, A., Baadshaug, O.H., Bélanger, G., Black, A.,
763 Brophy, C., Collins, R.P. and Čop, J., 2013. Ecosystem function enhanced by combining four functional types of
764 plant species in intensively managed grassland mixtures: a 3-year continental-scale field experiment. *Journal of*
765 *Applied Ecology*, 50, 365-375.

766

767 Fry, E.L., Manning, P., Allen, D.G., Hurst, A., Everwand, G., Rimmler, M. and Power, S.A., 2013. Plant
768 functional group composition modifies the effects of precipitation change on grassland ecosystem function. *PloS*
769 *One*, 8, p.e57027.

770

771 Forum for the Future of Agriculture (FFA) <http://www.forumforagriculture.com/>

772

773 F.R.A.N.Z.: www.franz-projekt.de

774

775 Gámez-Virués, S., Perović, D.J., Gossner, M.M., Börschig, C., Blüthgen, N., De Jong, H., Simons, N.K., Klein,
776 A.M., Krauss, J., Maier, G. and Scherber, C., et al. 2015. Landscape simplification filters species traits and
777 drives biotic homogenization. *Nature Communications*, 6, 8568.

778

779 Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M.C., Fröberg, M.,
780 Stendahl, J., Phillipson, C.D. and Mikusiński, G., 2013. Higher levels of multiple ecosystem services are found in
781 forests with more tree species. *Nature Communications*, 4, 1340.

782
783 Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Kremen, C.,
784 Carvalho, L.G., Harder, L.D., Afik, O. and Bartomeus, I., et al. 2013. Wild pollinators enhance fruit set of
785 crops regardless of honey bee abundance. *Science*, 339, 1608-1611.
786
787 Garibaldi, L.A., Carvalho, L.G., Leonhardt, S.D., Aizen, M.A., Blaauw, B.R., Isaacs, R., Kuhlmann, M.,
788 Kleijn, D., Klein, A.M., Kremen, C. and Morandin, L., et al. 2014. From research to action: enhancing crop yield
789 through wild pollinators. *Frontiers in Ecology and the Environment*, 12, 439-447.
790
791 Garibaldi, L.A., Bartomeus, I., Bommarco, R., Klein, A.M., Cunningham, S.A., Aizen, M.A., Boreux, V.,
792 Garratt, M.P., Carvalho, L.G., Kremen, C. and Morales, C.L., 2015. Trait matching of flower visitors and
793 crops predicts fruit set better than trait diversity. *Journal of Applied Ecology*, 52, 1436-1444.
794
795 Garibaldi, L., Pérez-Méndez, N., Garratt, M. P. D., Gemmill-Herren, B., Miguez, F. & Dicks, L. 2019: Policies
796 for ecological intensification of crop production. *Trends in Ecology and Evolution*.
797
798 Gardiner, M.M., Landis, D.A., Gratton, C., DiFonzo, C.D., O'neal, M., Chacon, J.M., Wayo, M.T., Schmidt,
799 N.P., Mueller, E.E. and Heimpel, G.E., 2009. Landscape diversity enhances biological control of an introduced
800 crop pest in the north-central USA. *Ecological Applications*, 19, 143-154.
801
802 Geertsema, W., Rossing, W.A., Landis, D.A., Bianchi, F.J., Van Rijn, P.C., Schaminée, J.H., Tschardtke, T. and
803 Van Der Werf, W., 2016. Actionable knowledge for ecological intensification of agriculture. *Frontiers in*
804 *Ecology and the Environment*, 14, 209-216.
805
806 Gérard, A., Wollni, M., Hölscher, D., Irawan, B., Sundawati, L., Teuscher, M. and Kreft, H., 2017. Oil-palm
807 yields in diversified plantations: Initial results from a biodiversity enrichment experiment in Sumatra, Indonesia.
808 *Agriculture, Ecosystems & Environment*, 240, 253-260.
809
810

811 Gossner, M.M., Lewinsohn, T.M., Kahl, T., Grassein, F., Boch, S., Prati, D., Birkhofer, K., Renner, S.C.,
812 Sikorski, J., Wubet, T., et al. 2016. Land-use intensification causes multitrophic homogenization of grassland
813 communities. *Nature* 540, 266.

814

815 Gould, I.J., Quinton, J.N., Weigelt, A., De Deyn, G.B. and Bardgett, R.D., 2016. Plant diversity and root traits
816 benefit physical properties key to soil function in grasslands. *Ecology Letters*, 19, 1140-1149.

817

818 Grab, H., Branstetter, M.G., Amon, N., Urban-Mead, K.R., Park, M.G., Gibbs, J., Blitzer, E.J., Poveda, K.,
819 Loeb, G. and Danforth, B.N., et al. 2019. Agriculturally dominated landscapes reduce bee phylogenetic
820 diversity and pollination services. *Science*, 363, 282-284.

821

822 Grace, J.B., Anderson, T.M., Seabloom, E.W., Borer, E.T., Adler, P.B., Harpole, W.S., Hautier, Y., Hillebrand,
823 H., Lind, E.M., Pärtel, M., et al. 2016. Integrative modelling reveals mechanisms linking productivity and plant
824 species richness. *Nature* 529, 390.

825

826 Grass, I., Loos, J., Baensch, S., Batáry, P., Librán-Embid, F., Ficiciyan, A., Klaus, F., Riechers, M., Rosa, J.,
827 Tiede, J. and Udy, K., 2019. Land-sharing/-sparing connectivity landscapes for ecosystem services and
828 biodiversity conservation. *People and Nature*. <https://doi.org/10.1002/pan3.21>

829

830 Grimm, V. and Wissel, C., 1997. Babel, or the ecological stability discussions: an inventory and analysis of
831 terminology and a guide for avoiding confusion. *Oecologia*, 109, 323-334.

832

833 Hass AL, Kormann UG, Tschardt T, Clough Y, Baillod AB, Sirami C, Fahrig L, Martin JL, Baudry J,
834 Bertrand C, Bosch J. Landscape configurational heterogeneity by small-scale agriculture, not crop diversity,
835 maintains pollinators and plant reproduction in western Europe. *Proceedings of the Royal Society B: Biological*
836 *Sciences*. 2018. 285, 20172242.

837

838 Hautier, Y., Isbell, F., Borer, E.T., Seabloom, E.W., Harpole, W.S., Lind, E.M., MacDougall, A.S., Stevens,
839 C.J., Adler, P.B., Alberti, J. and Bakker, J.D., et al. 2018. Local loss and spatial homogenization of plant
840 diversity reduce ecosystem multifunctionality. *Nature Ecology and Evolution*, 2, 50.

841

842 Hector, A., Schmid, B., Beierkuhnlein, C., Caldeira, M., Diemer, M., Dimitrakopoulos, P., Finn, J., Freitas, H.,
843 Giller, P., Good, J., et al. 1999. Plant diversity and productivity experiments in European grasslands. *Science*
844 286, 1123-1127.

845

846 Hill, R., Nates-Parra, G., Quezada-Euán, J.J.G., Buchori, D., LeBuhn, G., Maués, M.M., Pert, P.L., Kwapong,
847 P.K., Saeed, S., Breslow, S.J. and da Cunha, M.C., et al. 2019. Biocultural approaches to pollinator conservation.
848 *Nature Sustainability*, 2, 214.

849

850 [Hines et al .- this issue.](#)

851

852 Hooper, D.U., Chapin, F., Ewel, J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J., Lodge, D., Loreau, M.,
853 Naeem, S., et al. 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge.
854 *Ecological Monographs* 75, 3-35.

855

856 Hooper, D.U., Adair, E.C., Cardinale, B.J., Byrnes, J.E., Hungate, B.A., Matulich, K.L., Gonzalez, A., Duffy,
857 J.E., Gamfeldt, L. and O'Connor, M.I., 2012. A global synthesis reveals biodiversity loss as a major driver of
858 ecosystem change. *Nature*, 486, 105.

859

860 Huang, Y., Chen, Y., Castro-Izaguirre, N., Baruffol, M., Brezzi, M., Lang, A., Li, Y., Härdtle, W., von Oheimb,
861 G., Yang, X. and Liu, X., et al. 2018. Impacts of species richness on productivity in a large-scale subtropical
862 forest experiment. *Science*, 362, 80-83.

863

864 Hulvey, K.B., Hobbs, R.J., Standish, R.J., Lindenmayer, D.B., Lach, L., Perring, M.P., 2013. Benefits of tree
865 mixes in carbon plantings. *Nature Climate Change* 3, 869.

866

867 Huston, M. A. 1997. Hidden treatments in ecological experiments: re-evaluating the ecosystem function of
868 biodiversity. *Oecologia*, 110, 449-460.

869

870 Ingerslew, K.S. and Kaplan, I., 2018. Distantly related crops are not better rotation partners for tomato. *Journal*
871 *of Applied Ecology*, 55, 2506-2516.

872

873 Isbell, F., Reich, P.B., Tilman, D., Hobbie, S.E., Polasky, S. and Binder, S., 2013. Nutrient enrichment,
874 biodiversity loss, and consequent declines in ecosystem productivity. *Proceedings of the National Academy of*
875 *Sciences*, 110, 11911-11916.

876

877 Isbell, F., Tilman, D., Polasky, S. and Loreau, M., 2015. The biodiversity-dependent ecosystem service
878 debt. *Ecology Letters*, 18, 119-134.

879

880 Isbell, F., Craven, D., Connolly, J., Loreau, M., Schmid, B., Beierkuhnlein, C., Bezemer, T.M., Bonin, C.,
881 Bruelheide, H., De Luca, E., et al. 2015. Biodiversity increases the resistance of ecosystem productivity to
882 climate extremes. *Nature* 526, 574-577.

883

884 Isbell, F., Adler, P.R., Eisenhauer, N., Fornara, D., Kimmel, K., Kremen, C., Letourneau, D.K., Liebman, M.,
885 Polley, H.W., Quijas, S., 2017. Benefits of increasing plant diversity in sustainable agroecosystems. *Journal of*
886 *Ecology* 105, 871-879.

887

888 Iverson AL, Marín LE, Ennis KK, Gonthier DJ, Connor-Barrie BT, Remfert JL, Cardinale BJ, Perfecto I. Do
889 polycultures promote win-wins or trade-offs in agricultural ecosystem services? A meta-analysis. *Journal of*
890 *Applied Ecology*. 2014. 51, 1593-602.

891

892 Jackson, L.E., Pulleman, M.M., Brussaard, L., Bawa, K.S., Brown, G.G., Cardoso, I.M., De Ruiter, P.C., García-
893 Barrios, L., Hollander, A.D., Lavelle, P. and Ouédraogo, E., 2012. Social-ecological and regional adaptation of
894 agrobiodiversity management across a global set of research regions. *Global Environmental Change*, 22, 623-
895 639.

896

897 Kampmann, D., Lüscher, A., Konold, W. and Herzog, F., 2012. Agri-environment scheme protects diversity of
898 mountain grassland species. *Land Use Policy*, 29, 569-576.

899

900 King, E., Cavender-Bares, J., Balvanera, P., Mwampamba, T. and Polasky, S., 2015. Trade-offs in ecosystem
901 services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities. *Ecology and*
902 *Society*, 20.

903

904 Khalsa, J., Fricke, T., Weigelt, A. and Wachendorf, M., 2014. Effects of species richness and functional groups
905 on chemical constituents relevant for methane yields from anaerobic digestion: results from a grassland diversity
906 experiment. *Grass and Forage Science*, 69, .49-63.

907

908 Kirmer, A., Baasch, A. and Tischew, S., 2012. Sowing of low and high diversity seed mixtures in ecological
909 restoration of surface mined-land. *Applied Vegetation Science*, 15, 198-207.

910

911 Kirwan, L., Lüscher, A., Sebastià, M.T., Finn, J.A., Collins, R.P., Porqueddu, C., Helgadóttir, A., Baadshaug,
912 O.H., Brophy, C., Coran, C. and Dalmanndóttir, S., et al. 2007. Evenness drives consistent diversity effects in
913 intensive grassland systems across 28 European sites. *Journal of Ecology*, 95, 530-539.

914

915 Klein, A.M., Steffan–Dewenter, I. and Tscharntke, T., 2003. Fruit set of highland coffee increases with the
916 diversity of pollinating bees. *Proceedings of the Royal Society of London. Series B: Biological Sciences*, 270,
917 955-961.

918

919 Kleijn, D., Winfree, R., Bartomeus, I., Carvalheiro, L.G., Henry, M., Isaacs, R., Klein, A.M., Kremen, C.,
920 M'gonigle, L.K., Rader, R. and Ricketts, T.H., 2015. Delivery of crop pollination services is an insufficient
921 argument for wild pollinator conservation. *Nature Communications*, 6, 7414.

922

923 Kleijn, D., Bommarco, R., Fijen, T.P., Garibaldi, L.A., Potts, S.G. and van der Putten, W.H., 2018. Ecological
924 Intensification: Bridging the Gap between Science and Practice. *Trends in Ecology and Evolution*.

925

926 Knapp, S. and van der Heijden, M.G., 2018. A global meta-analysis of yield stability in organic and conservation
927 agriculture. *Nature Communications*, 9, 3632.

928

929 Koh, I., E. V. Lonsdorf, N. Williams, C. Brittain, R. Isaacs, J. Gibbs and T. H. Ricketts. 2016. Modeling the
930 status, trends, and impacts of wild bee abundance in the United States. *Proceedings of the National Academy of*
931 *Sciences* 113, 140-145.

932

933 Kollmann, J., Meyer, S.T., Bateman, R., Conradi, T., Gossner, M.M., de Souza Mendonça Jr, M., Fernandes,
934 G.W., Hermann, J.M., Koch, C., Müller, S.C. and Oki, Y., 2016. Integrating ecosystem functions into restoration
935 ecology—recent advances and future directions. *Restoration Ecology*, 24, 722-730.

936

937 Kormann, U., Rösch, V., Batáry, P., Tschardtke, T., Orci, K.M., Samu, F. and Scherber, C., 2015. Local and
938 landscape management drive trait-mediated biodiversity of nine taxa on small grassland fragments. *Diversity and*
939 *Distributions*, 21, 1204-1217.

940

941 Kremen, C. and Miles, A., 2012. Ecosystem services in biologically diversified versus conventional farming
942 systems: benefits, externalities, and trade-offs. *Ecology and Society*, 17.

943

944 Kremen, C. & Merenlender A.M. 2018. Landscapes that work for biodiversity and people. *Science*, 362,
945 eaau6020.

946

947 Landis, D.A. 2017. Designing agricultural landscapes for biodiversity-based ecosystem services. *Basic and*
948 *Applied Ecology*. 18, 1-12.

949

950 Larsen, T.H., Williams, N.M. and Kremen, C., 2005. Extinction order and altered community structure rapidly
951 disrupt ecosystem functioning. *Ecology letters*, 8, 538-547.

952

953 Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.P., Garden, D., Girel, J., Pellet, G. and Douzet, R., 2011.
954 Using plant functional traits to understand the landscape distribution of multiple ecosystem services. *Journal of*
955 *Ecology*, 99, 135-147.

956

957 Lefcheck, J.S., Byrnes, J.E., Isbell, F., Gamfeldt, L., Griffin, J.N., Eisenhauer, N., Hensel, M.J., Hector, A.,
958 Cardinale, B.J. and Duffy, J.E., 2015. Biodiversity enhances ecosystem multifunctionality across trophic levels
959 and habitats. *Nature Communications*, 6, 6936.

960

961 Lange, M., Eisenhauer, N., Sierra, C.A., Bessler, H., Engels, C., Griffiths, R.I., Mellado-Vázquez, P.G., Malik,
962 A.A., Roy, J., Scheu, S. and Steinbeiss, S., et al. 2015. Plant diversity increases soil microbial activity and soil
963 carbon storage. *Nature Communications*, 6, 6707.

964

965 Larsen, T.H., Williams, N.M. & Kremen, C. 2005. Extinction order and altered community structure rapidly
966 disrupt ecosystem functioning. *Ecology Letters*, 8, 538–547

967

968 Lavorel, S., Bayer, A., Bondeau, A., Lautenbach, S., Ruiz-Frau, A., Schulp, N., Seppelt, R., Verburg, P., van
969 Teeffelen, A., Vannier, C. and Arneith, A., 2017. Pathways to bridge the biophysical realism gap in ecosystem
970 services mapping approaches. *Ecological Indicators*, 74, 241-260.

971

972 Leibold, M.A., Chase, J.M. and Ernest, S.M., 2017. Community assembly and the functioning of ecosystems:
973 how metacommunity processes alter ecosystems attributes. *Ecology*, 98, 909-919.

974

975 Lepš, J. 2004. What do the biodiversity experiments tell us about consequences of plant species loss in the real
976 world? *Basic and Applied Ecology*, 5, 529-534.

977

978 Lichtenberg, E.M., Kennedy, C.M., Kremen, C., Batáry, P., Berendse, F., Bommarco, R., Bosque-Pérez, N.A.,
979 Carvalheiro, L.G., Snyder, W.E., Williams, N.M. and Winfree, R., 2017. A global synthesis of the effects of
980 diversified farming systems on arthropod diversity within fields and across agricultural landscapes. *Global
981 Change Biology*, 23, 4946-4957.

982

983 Lindborg, R., L. J. Gordon, R. Malinga, J. Bengtsson, G. Peterson, R. Bommarco, L. Deutsch, A. Gren, M.
984 Rundlöf, and H. G. Smith. 2017. How spatial scale shapes the generation and management of multiple
985 ecosystem services. *Ecosphere* 8, e01741. 10.1002/ecs2.1741

986

987 Loreau, M., Hector, A., 2001. Partitioning selection and complementarity in biodiversity experiments. *Nature*
988 413, 548-548.

989

990 Loreau, M., Naeem, S., Inchausti, P., Bengtsson, J., Grime, J.P., Hector, A., Hooper, D.U., Huston, M.A.,
991 Raffaelli, D., Schmid, B., Tilman, D., Wardle, D.A., 2001. Biodiversity and ecosystem functioning: Current
992 knowledge and future challenges. *Science* 294, 804-808.

993

994 Loreau, M., Mouquet, N. and Gonzalez, A., 2003. Biodiversity as spatial insurance in heterogeneous landscapes.
995 Proceedings of the National Academy of Sciences, 100, 12765-12770.
996
997 Lundholm, J., MacIvor, J.S., MacDougall, Z. and Ranalli, M., 2010. Plant species and functional group
998 combinations affect green roof ecosystem functions. PloS one, 5, e9677.
999
1000 Lyons, K.G. and Schwartz, M.W., 2001. Rare species loss alters ecosystem function–invasion resistance.
1001 Ecology Letters, 4, 358-365.
1002
1003 Mace, G.M., Norris, K. and Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship.
1004 Trends in Ecology and Evolution, 27, 19-26.
1005
1006 Maestre, F.T., Quero, J.L., Gotelli, N.J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M.,
1007 Bowker, M.A., Soliveres, S., Escolar, C., 2012. Plant species richness and ecosystem multifunctionality in global
1008 drylands. Science 335, 214-218.
1009
1010 Manning, P., Newington, J.E., Robson, H.R., Saunders, M., Eggers, T., Bradford, M.A., Bardgett, R.D.,
1011 Bonkowski, M., Ellis, R.J., Gange, A.C., 2006. Decoupling the direct and indirect effects of nitrogen deposition
1012 on ecosystem function. Ecology Letters 9, 1015-1024.
1013
1014 Manning, P., Vries, F.T., Tallwin, J.R., Smith, R., Mortimer, S.R., Pilgrim, E.S., Harrison, K.A., Wright, D.G.,
1015 Quirk, H., Benson, J. and Shipley, B., et al. 2015. Simple measures of climate, soil properties and plant traits
1016 predict national-scale grassland soil carbon stocks. Journal of Applied Ecology, 52, 1188-1196.
1017
1018 Manning, P., Plas, F., Soliveres, S., Allan, E., Maestre, F.T., Mace, G., Whittingham, M.J., Fischer, M., 2018.
1019 Redefining ecosystem multifunctionality. Nature Ecology and Evolution 2, 427.
1020
1021 Maron, J.L., Marler, M., Klironomos, J.N., Cleveland, C.C., 2011. Soil fungal pathogens and the relationship
1022 between plant diversity and productivity. Ecology Letters 14, 36-41.
1023

1024 Martín-López, B., Iniesta-Arandia, I., García-Llorente, M., Palomo, I., Casado-Arzuaga, I., Del Amo, D.G.,
1025 Gómez-Baggethun, E., Oteros-Rozas, E., Palacios-Agundez, I., Willaarts, B. and González, J.A., 2012.
1026 Uncovering ecosystem service bundles through social preferences. *PLoS One*, 7, e38970.
1027
1028 Maskell, L. C., Crowe, A. , Dunbar, M. J., Emmett, B. , Henrys, P. , Keith, A. M., Norton, L. R., Scholefield, P. ,
1029 Clark, D. B., Simpson, I. C., Smart, S. M., Clough, Y. 2013. Exploring the ecological constraints to multiple
1030 ecosystem service delivery and biodiversity. *Journal of Applied Ecology*, 50, 561-571.
1031
1032 Meyer, S.T., Ptacnik, R., Hillebrand, H., Bessler, H., Buchmann, N., Ebeling, A., Eisenhauer, N., Engels, C.,
1033 Fischer, M., Halle, S. and Klein, A.M., et al. 2018. Biodiversity–multifunctionality relationships depend on
1034 identity and number of measured functions. *Nature Ecology and Evolution*, 2, 44.
1035
1036 Mori, A.S. Isbell, I & Seidl, R. 2018. β -Diversity, Community Assembly, and Ecosystem Functioning. *Trends in*
1037 *Ecology and Evolution* 33, 549-564.
1038
1039 Nyfeler, D. , Huguenin-Elie, O. , Suter, M., Frossard, E. , Connolly, J. and Lüscher, A. 2009. Strong mixture
1040 effects among four species in fertilized agricultural grassland led to persistent and consistent transgressive
1041 overyielding. *Journal of Applied Ecology*, 46, 683-691.
1042
1043 Oliver, T.H., Heard, M.S., Isaac, N.J., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme,
1044 C.D.L., Petchey, O.L., 2015. Biodiversity and resilience of ecosystem functions. *Trends in Ecology and*
1045 *Evolution* 30, 673-684.
1046
1047 Otway, S.J., Hector, A., Lawton, J.H., 2005. Resource dilution effects on specialist insect herbivores in a
1048 grassland biodiversity experiment. *Journal of Animal Ecology* 74, 234-240.
1049
1050 Pan, Q., Tian, D., Naeem, S., Auerswald, K., Elser, J.J., Bai, Y., Huang, J., Wang, Q., Wang, H., Wu, J. and
1051 Han, X., 2016. Effects of functional diversity loss on ecosystem functions are influenced by compensation.
1052 *Ecology*, 97, 2293-2302.
1053

1054 Peeters, A., Beaufoy, G., Canals, R.M., de Vlieghe, A., Huyghe, C., Isselstein, J., Jones, G., Kessler W.,
1055 Kirilov, A., Mosquera-Losada, M.R., et al. 2014. Grassland term definitions and and classifications adapted to
1056 the delivery of European grassland-based systems. *Grassland Science in Europe* 19, 743-750.
1057
1058 van der Plas F, Manning P, Soliveres S, Allan E, Scherer-Lorenzen M, Verheyen K, Wirth C, Zavala MA, ,
1059 Ampoorter E, Baeten L, Barbaro L et al. (2016) Biotic homogenization can decrease landscape-scale forest
1060 multifunctionality. *Proceedings of the National Academy of Sciences*, 113, 3557-3562
1061
1062 van der Plas, F., Ratcliffe, S., Ruiz-Benito, P., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M.A.,
1063 Ampoorter, E., Baeten, L., Barbaro, L. and Bastias, C.C., et al. 2018. Continental mapping of forest ecosystem
1064 functions reveals a high but unrealised potential for forest multifunctionality. *Ecology Letters*, 21, 31-42.
1065
1066 van der Plas, F., 2019a. Biodiversity and ecosystem functioning in naturally assembled communities. *Biological*
1067 *Reviews*.
1068
1069 van der Plas, F., Allan, E., Fischer, M., Alt, F., Arndt, H., Binkenstein, J., Blaser, S., Blüthgen, N., Böhm, S.,
1070 Hölzel, N. and Klaus, V.H., et al. 2019b. Towards the development of general rules describing landscape
1071 heterogeneity–multifunctionality relationships. *Journal of Applied Ecology*, 56, 168-179.
1072
1073 Pretzsch, H. and Schütze, G., 2009. Transgressive overyielding in mixed compared with pure stands of Norway
1074 spruce and European beech in Central Europe: evidence on stand level and explanation on individual tree level.
1075 *European Journal of Forest Research*, 128, 183-204.
1076
1077 Van der Putten, W.H., Mortimer, S.R., Hedlund, K., Van Dijk, C., Brown, V.K., Lepä, J., Rodriguez-Barrueco,
1078 C., Roy, J., Len, T.D., Gormsen, D. and Korthals, G.W., 2000. Plant species diversity as a driver of early
1079 succession in abandoned fields: a multi-site approach. *Oecologia*, 124, 91-99.
1080
1081 Raseduzzaman, M. and Jensen, E.S., 2017. Does intercropping enhance yield stability in arable crop production?
1082 A meta-analysis. *European Journal of Agronomy*, 91, 25-33.
1083

1084 Rader, R., Birkhofer, K., Schmucki, R., Smith, H.G., Stjernman, M. and Lindborg, R., 2014. Organic farming
1085 and heterogeneous landscapes positively affect different measures of plant diversity. *Journal of Applied*
1086 *Ecology*, 51, 1544-1553.

1087

1088 Redlich, S., Martin, E.A. and Steffan-Dewenter, I., 2018. Landscape-level crop diversity benefits biological pest
1089 control. *Journal of Applied Ecology*, 55, 2419-2428.

1090

1091 Reiss, E.R., Drinkwater, L.E., 2018. Cultivar mixtures: a meta-analysis of the effect of intraspecific diversity on
1092 crop yield. *Ecological Applications*, 28, 62-77.

1093

1094 Rosa J, Loos J, Musshoff O, Tschardtke T (2019) Ecological-economic trade-offs of Diversified farming
1095 systems - a review. *Ecological Economics*.

1096

1097 Rösch, V., Tschardtke, T., Scherber, C. and Batáry, P., 2015. Biodiversity conservation across taxa and
1098 landscapes requires many small as well as single large habitat fragments. *Oecologia*, 179, 209-222.

1099

1100 Roscher, C., Schumacher, J., Baade, J., Wilcke, W., Gleixner, G., Weisser, W.W., Schmid, B., Schulze, E.-D.,
1101 2004. The role of biodiversity for element cycling and trophic interactions: an experimental approach in a
1102 grassland community. *Basic and Applied Ecology* 5, 107-121.

1103

1104 Roschewitz, I., Gabriel, D., Tschardtke, T., Thies, C.. 2005. The effects of landscape complexity on arable weed
1105 species diversity in organic and conventional farming. *Journal of Applied Ecology*. 2005, 42, 873-82.

1106

1107 RSPB Hope Farm [https://www.rspb.org.uk/our-work/conservation/conservation-and-](https://www.rspb.org.uk/our-work/conservation/conservation-and-sustainability/farming/hope-farm/)
1108 [sustainability/farming/hope-farm/](https://www.rspb.org.uk/our-work/conservation/conservation-and-sustainability/farming/hope-farm/)

1109

1110 Samnegard, U., Alins, G., Boreux, V., Bosch, J., García, D., Happe, A.-K., Klein, A.M., Miñarro, M, Mody, K.,
1111 Porcel, M., et al. 2018. Management trade-offs on ecosystem services in apple orchards across Europe: direct
1112 and indirect effects of organic production. *Journal of Applied Ecology*.

1113

1114 Scherber, C., Eisenhauer, N., Weisser, W.W., Schmid, B., Voigt, W., Fischer, M., Schulze, E.-D., Roscher, C.,
1115 Weigelt, A., Allan, E., et al. 2010. Bottom-up effects of plant diversity on multitrophic interactions in a
1116 biodiversity experiment. *Nature* 468, 553.
1117
1118 Schnitzer, S.A., Klironomos, J.N., HilleRisLambers, J., Kinkel, L.L., Reich, P.B., Xiao, K., Rillig, M.C., Sikes,
1119 B.A., Callaway, R.M., Mangan, S.A., 2011. Soil microbes drive the classic plant diversity–productivity pattern.
1120 *Ecology* 92, 296-303.
1121
1122 Schmid, B., Hector, A., Huston, M.A., Inchausti, P., Nijs, I., Leadley, P.W. and Tilman, D., 2002. The design
1123 and analysis of biodiversity experiments. *Biodiversity and ecosystem functioning: synthesis and perspectives*,
1124 pp.61-75.
1125
1126 Schmid, B. and Hector, A., 2004. The value of biodiversity experiments. *Basic and Applied Ecology*, 5, 535-
1127 542.
1128
1129 Schulze, E.-D., Mooney, H.A., 1994. Ecosystem function of biodiversity: a summary. *Biodiversity and*
1130 *ecosystem function*. Springer, pp. 497-510.
1131
1132 Smith, M.D., Knapp, A.K., 2003. Dominant species maintain ecosystem function with non-random species loss.
1133 *Ecology Letters*, 6, 509-517.
1134
1135 Smith, R., Shiel, R., Bardgett, R.D., Millward, D., Corkhill, P., Evans, P., Quirk, H., Hobbs, P., Kometa, S.,
1136 2008. Long-term change in vegetation and soil microbial communities during the phased restoration of
1137 traditional meadow grassland. *Journal of Applied Ecology* 45, 670-679.
1138
1139 Soliveres, S., Manning, P., Prati, D., Gossner, M.M., Alt, F., Arndt, H., Baumgartner, V., Binkenstein, J.,
1140 Birkhofer, K., Blaser, S., et al. 2016a. Locally rare species influence grassland ecosystem multifunctionality.
1141 *Phil. Trans. R. Soc. B* 371, 20150269.
1142

1143 Soliveres, S., Van Der Plas, F., Manning, P., Prati, D., Gossner, M.M., Renner, S.C., Alt, F., Arndt, H.,
1144 Baumgartner, V., Binkenstein, J., et al. 2016b. Biodiversity at multiple trophic levels is needed for ecosystem
1145 multifunctionality. *Nature* 536, 456.
1146
1147 Srivastava, D.S., Vellend, M., 2005. Biodiversity-ecosystem function research: is it relevant to conservation?
1148 *Annual Review of Ecology Evolution and Systematics* 36, 267-294.
1149
1150 Stein, C., Auge, H., Fischer, M., Weisser, W.W. and Prati, D., 2008. Dispersal and seed limitation affect
1151 diversity and productivity of montane grasslands. *Oikos*, 117, 1469-1478.
1152
1153 Storkey, J., Döring, T., Baddeley, J., Collins, R., Roderick, S., Jones, H., Watson, C., 2015. Engineering a plant
1154 community to deliver multiple ecosystem services. *Ecological Applications* 25, 1034-1043.
1155
1156 Suding, K.N., Ashton, I.W., Bechtold, H., Bowman, W.D., Mobley, M.L., Winkleman, R., 2008. Plant and
1157 microbe contribution to community resilience in a directionally changing environment.
1158 *Ecological Monographs*, 78, 313-329.
1159
1160 Suter, M., Connolly, J., Finn, J.A., Loges, R., Kirwan, L., Sebastià, M.T. and Lüscher, A., 2015. Nitrogen yield
1161 advantage from grass–legume mixtures is robust over a wide range of legume proportions and environmental
1162 conditions. *Global Change Biology*, 21, 2424-2438.
1163
1164 Tallis, H., 2011. *Natural capital: theory and practice of mapping ecosystem services*. Oxford University Press.
1165
1166 Teuscher, M., Gérard, A., Brose, U., Buchori, D., Clough, Y., Ehbrecht, M., Hölscher, D., Irawan, B.,
1167 Sundawati, L., Wollni, M. & Kreft, H. 2016. Experimental Biodiversity Enrichment in Oil-Palm-Dominated
1168 Landscapes in Indonesia. *Frontiers of Plant Science*, 7, 1538.
1169
1170 Tilman, D., Wedin, D., Knops, J., 1996. Productivity and sustainability influenced by biodiversity in grassland
1171 ecosystems. *Nature* 379, 718.
1172

1173 Tilman, D., Reich, P.B., Knops, J., Wedin, D., Mielke, T., Lehman, C., 2001. Diversity and productivity in a
1174 long-term grassland experiment. *Science* 294, 843-845.
1175
1176 Tilman, D., Hill, J. and Lehman, C., 2006. Carbon-negative biofuels from low-input high-diversity grassland
1177 biomass. *Science*, 314, 1598-1600.
1178
1179 Tschardtke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I. and Thies, C., 2005. Landscape perspectives on
1180 agricultural intensification and biodiversity–ecosystem service management. *Ecology letters*, 8, 857-874.
1181
1182 Tschardtke, T., Tylianakis, J.M., Rand, T.A., Didham, R.K., Fahrig, L., Batáry, P., Bengtsson, J., Clough, Y.,
1183 Crist, T.O., Dormann, C.F. and Ewers, R.M., 2012. Landscape moderation of biodiversity patterns and
1184 processes-eight hypotheses. *Biological Reviews*, 87, 661-685.
1185
1186 Tschardtke, T., Karp, D.S., Chaplin-Kramer, R., Batáry, P., DeClerck, F., Gratton, C., Hunt, L., Ives, A.,
1187 Jonsson, M., Larsen, A. and Martin, E.A., 2016. When natural habitat fails to enhance biological pest control–
1188 Five hypotheses. *Biological Conservation*, 204, 449-458.
1189
1190 Teuscher, M., Gérard, A., Brose, U., Buchori, D., Clough, Y., Ehbrecht, M., Hölscher, D., Irawan, B.,
1191 Sundawati, L., Wollni, M. and Kreft, H., 2016. Experimental biodiversity enrichment in oil-palm-dominated
1192 landscapes in Indonesia. *Frontiers in Plant Science*, 7, 1538.
1193
1194 UK National Ecosystem Assessment, 2011. The UK national ecosystem assessment: synthesis of the key
1195 findings.
1196
1197 Vandermeer, J. H. (1992). *The ecology of intercropping*. Cambridge University Press.
1198
1199 Vellend, M., Baeten, L., Myers-Smith, I.H., Elmendorf, S.C., Beauséjour, R., Brown, C.D., De Frenne, P.,
1200 Verheyen, K. and Wipf, S., 2013. Global meta-analysis reveals no net change in local-scale plant biodiversity
1201 over time. *Proceedings of the National Academy of Sciences*, 110, 19456-19459.
1202
1203 **Vogel et al 2019, this issue- to be added**

1204

1205 Verheyen, K., Vanhellemont, M., Auge, H., Baeten, L., Baraloto, C., Barsoum, N., Bilodeau-Gauthier, S.,
1206 Bruelheide, H., Castagneyrol, B., Godbold, D. and Haase, J., 2016. Contributions of a global network of tree
1207 diversity experiments to sustainable forest plantations. *Ambio*, 45, 29-41.

1208

1209 Walter, A., Finger, R., Huber, R. and Buchmann, N., 2017. Opinion: Smart farming is key to developing
1210 sustainable agriculture. *Proceedings of the National Academy of Sciences*, 114, 6148-6150.

1211

1212 Wardle, D.A., 2016. Do experiments exploring plant diversity–ecosystem functioning relationships inform how
1213 biodiversity loss impacts natural ecosystems? *Journal of Vegetation Science* 27, 646-653.

1214

1215 Weidlich, E.W., von Gillhaussen, P., Max, J.F., Delory, B.M., Jablonowski, N.D., Rascher, U. and Temperton,
1216 V.M., 2018. Priority effects caused by plant order of arrival affect below-ground productivity. *Journal of*
1217 *Ecology*, 106, 774-780.

1218

1219 Weisser WW, Roscher C, Meyer ST, Ebeling A, Luo G, Allan E, Beßler H, Barnard R, Buchmann N, Buscot F,
1220 Engels C, et al. (2017) Biodiversity effects on ecosystem functioning in a 15-year grassland experiment:
1221 Patterns, mechanisms, and open questions. *Basic and Applied Ecology* 23, 1-73.

1222

1223 Weidlich, E.W., von Gillhaussen, P., Max, J.F., Delory, B.M., Jablonowski, N.D., Rascher, U. and Temperton,
1224 V.M., 2018. Priority effects caused by plant order of arrival affect below-ground productivity. *Journal of*
1225 *Ecology*, 106, 774-780.

1226

1227 Werling, B.P., T.L. Dickson, R. Isaacs, H. Gaines, C. Gratton, K.L. Gross, H. Liere, C.M. Malmstrom, T.D.
1228 Meehan, L. Ruan, B.A. et al. 2014. Perennial grasslands enhance biodiversity and multiple ecosystem services
1229 in bioenergy landscapes. *Proceedings of the National Academy of Sciences* 111, 41652–1657.

1230

1231 Wilson, J. B., Peet, R. K., Dengler J. & Pärtel, M. 2012. Plant species richness: the world records. – *Journal of*
1232 *Vegetation Science* 23, 796–802.

1233

1234 Winfree, R., Reilly, J.R., Bartomeus, I., Cariveau, D.P., Williams, N.M., Gibbs, J., 2018. Species turnover
1235 promotes the importance of bee diversity for crop pollination at regional scales. *Science* 359, 791-793.
1236

1237 Winfree, R., W Fox, J., Williams, N.M., Reilly, J.R., Cariveau, D.P., 2015. Abundance of common species, not
1238 species richness, drives delivery of a real-world ecosystem service. *Ecology Letters* 18, 626-635.
1239

1240 Wright, A.J., Ebeling, A., De Kroon, H., Roscher, C., Weigelt, A., Buchmann, N., Buchmann, T., Fischer, C.,
1241 Hacker, N., Hildebrandt, A. and Leimer, S., et al. 2015. Flooding disturbances increase resource availability and
1242 productivity but reduce stability in diverse plant communities. *Nature Communications*, 6, 6092.
1243

1244 Wright, A.J., Wardle, D.A., Callaway, R., Gaxiola, A., 2017. The overlooked role of facilitation in biodiversity
1245 experiments. *Trends in Ecology and Evolution* 32, 383-390.
1246

1247 Yang, Y., Tilman, D., Furey, G., & Lehman, C. 2019. Soil carbon sequestration accelerated by restoration of
1248 grassland biodiversity. *Nature Communications*, 10, 718.
1249

1250 Yang, Y., Tilman, D., Lehman, C. and Trost, J.J., 2018. Sustainable intensification of high-diversity biomass
1251 production for optimal biofuel benefits. *Nature Sustainability*, 1, 686.
1252

1253 Zavaleta, E.S. and Hulvey, K.B., 2004. Realistic species losses disproportionately reduce grassland resistance to
1254 biological invaders. *Science*, 306, 1175-1177.
1255

1256 Zhu, Y., Chen, H., Fan, J., Wang, Y., Li, Y., Chen, J., Fan, J., Yang, S., Hu, L., Leung, H., 2000. Genetic
1257 diversity and disease control in rice. *Nature*, 406, 718.
1258

1259 Zhu, J., Jiang, L. and Zhang, Y. 2016. Relationships between Functional Diversity and Aboveground Biomass
1260 Production in the Northern Tibetan Alpine Grasslands. *Scientific Reports* 6, 34105.
1261

1262 Zobel, K., Zobel, M., & Rosén, E. 1994. An experimental test of diversity maintenance mechanisms, by a
1263 species removal experiment in a species-rich wooded meadow. *Folia Geobotanica et Phytotaxonomica*, 29, 449-
1264 457.

1265
1266
1267
1268
1269
1270
1271
1272
1273
1274
1275
1276
1277
1278
1279
1280
1281
1282
1283
1284
1285
1286
1287
1288
1289
1290
1291

Zuppinger-Dingley, D., Schmid, B., Petermann, J.S., Yadav, V., De Deyn, G.B. and Flynn, D.F., 2014. Selection for niche differentiation in plant communities increases biodiversity effects. *Nature*, 515, 108.

Table 1. Research required to enable the real-world application of BEF research

| Research need and approach | Potential benefit to | Examples or foundational |
|----------------------------|----------------------|--------------------------|
|----------------------------|----------------------|--------------------------|

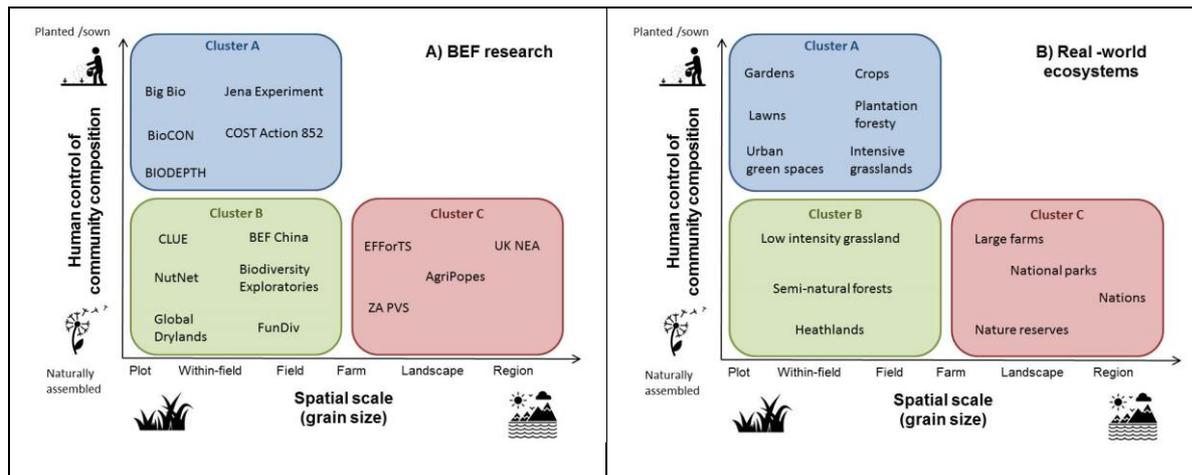
| | transfer | studies |
|---|---|--|
| Cluster A | | |
| Identify mechanistic general rules governing complementary species combinations in existing biodiversity experiments | Suggested combinations of species for restoration, intercropping and crop rotation, mixed plantations etc. | Zuppinger-Dingley et al. (2014) Brooker et al. (2015) |
| Demonstrate the biodiversity-multifunctionality relationship in sown or planted ecosystems, e.g. by identifying mixtures that provide multiple desired services | Could be used to design multifunctional species mixtures that provide benefits to a range of stakeholder groups | Baeten et al. (2019) Finn et al. (2013) |
| Compare multispecies mixtures to the high performing species-poor systems of current management | Without realistic comparison to current management alternative option will not be adopted | Binder et al (2018) |
| Perform BEF experiments with species pools that contain potentially useful and manageable species (e.g. self-sustaining mixtures) | High performing mixtures identified can be managed in a cost-effective manner | Kirwan et al (2007) Finn et al (2013) |
| Generate measures of stability that are relevant to managers | To show relationship between biodiversity and the stability sought by stakeholders | Donohue et al (2016) Oliver et al. (2015) |
| Demonstrate the cost effectiveness of multispecies mixtures compared to existing management and develop | Unless clear benefits are demonstrated diversification may not be adopted | Finger & Buchmann (2015) Blaauw & Isaacs (2014) |

| | | |
|--|---|---|
| technology that increases this (e.g. multicrop harvesters) | | |
| Cluster B | | |
| Form general predictions of how biodiversity and other drivers of ecosystem function changes in response to global change drivers | Accurate and general estimates and predictions of biodiversity loss are the foundation of accurate and general assessments of their impacts | Bjorkman et al (2018) Grace et al (2016) |
| Develop mechanistic understanding of biodiversity in real world systems, e.g. by using new quantitative tools to disentangle biodiversity effects | Would increase confidence in correlational BEF relationships and allow their causes to be understood | Grace et al (2016) |
| Systematically assess the relative role of alpha and beta diversity, functional composition, abundance and other covariates including abiotic factors and understand the feedbacks and relationships between these drivers | Would lead to more precise estimates of the relative role of biodiversity in semi-natural systems and its relationship with other factors | Allan et al (2015) Winfree et al (2015) van der Plas et al (2016) |
| Establish a new generation of experiments that varies the above factors, across realistic gradients | Would allow causation to be inferred for the above relationships | Smith & Knapp (2003) Manning et al (2006) |
| Assess the role of biodiversity in species rich communities, including that of rare species | Most diversity loss occurs between high and intermediate levels and rare | Soliveres et al. (2016b) Klein et al (2003) Lyons & Schwartz (2001) |

| | | |
|--|---|---|
| | species are more likely to be lost | |
| Provide statistical estimates of where different components of biodiversity play their greatest role and test these estimates | Can be used in regional and global assessments and projections of the expected impacts of biodiversity loss | van der Plas (2019a) |
| Explore the BEF relationship within the context of ecosystem restoration, and link this to community assembly mechanisms | The restoration of semi-natural habitats may be more effective if a high diversity of species is used | Bullock et al. (2007) Weidlich et al. (2018) |
| Cluster C | | |
| Understand the strength and role of mechanisms linking biodiversity to ecosystem function at spatial and temporal scales (e.g. species matching to site conditions, dispersal processes) | Biodiversity may play a different role at large scales to that established in experiments | Loreau et al (2003) Mori et al (2018) |
| Upscale ecosystem functions to large scales and link these to ecosystem services | Would allow the relationship between biodiversity, ecosystem functions and ecosystem services to be evaluated at management relevant scales | Clough et al (2016) Lindborg et al (2017) LeClec'h et al. (subm.) |
| Use upscaled measures to understand which taxa drive ecosystem services and disservices at landscape scales, and what factors | Would allow important ecosystem service providers to be identified and managed appropriately | Van der Plas et al (2018) Winfree et al (2018) Grass et al (2019) |

| | | |
|---|--|--|
| drive the diversity of these taxa | | |
| Evaluate the role of biodiversity in driving landscape multifunctionality of ecosystem services (via upscaled measures) | Would allow the impact of biodiversity on a range of stakeholders and wider society to be communicated | Van der Plas et al (2018) Manning et al (2018) |
| Knowledge exchange (all clusters) | | |
| Disseminate research findings effectively (e.g. via web tools and demonstration sites). | Non-academic approaches are required for BEF research findings to reach potential end-users users | Activities of: Forum for the Future of Agriculture (FFA) European Landowners Organisation (ELO) F.R.A.N.Z. Conservation evidence website RSPB Hope Farm |
| Work in collaboration with stakeholders to collect information on which ecosystem services are desired, at which different temporal and spatial scales, and their relative importance | This could inform applied BEF research, ensuring that it meets the needs of potential end-users | Geertstema et al (2016) Walter et al. (2017) |

1292



1293

1294 **Figure 1.** Clusters of BEF research and their relation to real world ecosystems. a) selected research
 1295 projects, b) selected ‘real-world’ ecosystems. Note that, as spatial scale increases, the user of
 1296 research findings changes from individual local scale managers to governments and
 1297 institutions and the form of transfer changes from management practice recommendations to
 1298 policy change, though these are clearly interrelated. Example references for the studies shown
 1299 are: Jena experiment (Weisser et al 2017), BigBio (Tilman et al. 2001), BioCON (Reich et al.
 1300 2001), COST Action 852 (Kirwan et al 2007), BIODEPTH (Hector 1999), BEF-China
 1301 (Huang et al. 2018), CLUE (van der Putten et al. 2000), NutNet (Grace et al., 2016),
 1302 Biodiversity Exploratories (Allan et al. 2015), Global Drylands (Maestre et al 2012), FunDiv
 1303 (Van der Plas et al 2016), EFForTS (Teuscher et al. 2016), AgriPopes (Emmerson et al.
 1304 2016), ZA PVS (Bretagnolle et al 2018), UKNEA National Ecosystem Assessment (2011).

1305



1306

1307 **Figure 2.** Swiss grassland diversification. In Switzerland species rich semi-natural grasslands
1308 (left) can decline to a more species-poor state (right) if fertilized and mown frequently. To
1309 counteract this loss many species rich sites are maintained via agri-environment policy
1310 schemes (Kampmann et al. 2012) and Swiss researchers have developed diversified seed
1311 mixtures suitable for a wide range of conditions that have been adopted by many Swiss
1312 farmers (Agrarforschung Schweiz 2019). We postulate that this adoption is likely to be
1313 attributable to a range of factors including: a strong cultural valuation of grassland, a clear
1314 mandate of agriculture to manage sustainably (in Swiss Constitution, article 104), generous
1315 agri-environment compensation schemes for a range of grassland types, and a strong focus on
1316 applied grassland research that has investigated which mixtures work over different time
1317 horizons (e.g. annual to permanent) and environmental conditions (moisture and elevational
1318 gradients) (e.g. Suter et al. 2015). Finally, there is effective communication from both
1319 researchers (e.g. Agroscope) and the Swiss grassland society (AGFF, 2019), which contains
1320 many farmers as members. Future BEF transfer work could investigate the role of such
1321 factors in successful transfer. Photo credits Peter Manning.