Transferring biodiversity-ecosystem function research to the management of 'real-world' ecosystems

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Abstract

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

1. Introduction

Widespread concerns over the consequences of global biodiversity loss led to an explosion of ecological research in the early 1990s into the relationship between biodiversity and the functioning of ecosystems (hereafter BEF research) (Schulze and Mooney, 1994; Loreau et al., 2001;

Hooper et al., 2005; Eisenhauer et al., 2019 this issue; Hines et al., 2019 this issue). Historically, most work in this field has been conducted in experimental settings, especially in grasslands, where extinction is simulated by randomly assembling plant communities differing in species and functional richness and where other environmental drivers of ecosystem function are controlled for (Hector et al., 1999; Tilman et al., 2001; Weisser et al., 2017). While this work has led to several robust conclusions regarding the form of biodiversity-function relationships and the mechanisms that drive them (Cardinale et al., 2012), there remain doubts regarding the capacity for experimental BEF research to inform the management of biodiversity and ecosystem functions and services in the 'real world' (i.e. ecosystems with communities that have not been experimentally manipulated) (Eisenhauer et al., 2016; Huston, 1997; Lepš, 2004; Srivastava and Vellend, 2005; Wardle, 2016). Much of this debate concerns the design of biodiversity experiments, which were established to investigate if biodiversity *could* affect function, and via what mechanisms (Loreau and Hector, 2001; Schmid et al., 2002; Tilman et al., 1996).

A more recent generation of BEF research has been conducted in non-experimental and naturally assembled real–world ecosystems such as natural and semi–natural (hereafter semi–natural) drylands, grasslands and forests (e.g. Maestre et al., 2012; Grace et al., 2016; Van Der Plas et al., 2016; Duffy et al., 2017; Fanin et al., 2018; Hautier et al., 2018, van der Plas, 2019). As they are performed in naturally assembled communities, shaped by both environmental drivers and global change factors, these studies are correlational and tend to rely upon statistical controls, thus limiting confident inference about the functional consequences of biodiversity loss in these systems. Removal experiments can help overcome this issue but, to date, relatively few have been conducted (Dıaz et al., 2003; Fanin et al., 2018; Fry et al., 2013). While a lack of confident inference may limit transfer, many other knowledge gaps also limit the transferability of BEF research. For example, there is little consensus regarding on how strongly biodiversity loss affects ecosystem functioning, relative to other drivers (Duffy et al., 2017; Hooper et al., 2012; Srivastava and Vellend, 2005; van der Plas, 2019). Moreover, the functional consequences of the non–random extinction which occurs in semi–natural ecosystems have largely been estimated from correlational studies (Larsen et al., 2005; Duffy et al., 2017; van der Plas, 2019, but see Lyons and Schwartz, 2001 and Zavaleta and Hulvey, 2004). Further challenges in the knowledge transfer and application of BEF research emerge from a lack of information regarding the social and economic barriers to conserving

biodiversity and promoting diversification (Fazey et al., 2013; Rosa–Schleich et al., 2019). Filling these knowledge gaps would help in providing reliable evidence to inform the management of the world's ecosystems, e.g., via the Intergovernmental Science-Policy Panel on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2015; Díaz et al., 2018).

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

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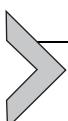
2. Small-grain and highly-controlled experiments (Cluster A)

Since the mid 90's, >600 experiments have been established to explore the causal relationship between biodiversity and ecosystem functioning (Cardinale et al., 2012), typically under field conditions (e.g. Hector et al., 1999; Roscher et al., 2004; Tilman et al., 1996). The primary goal of these
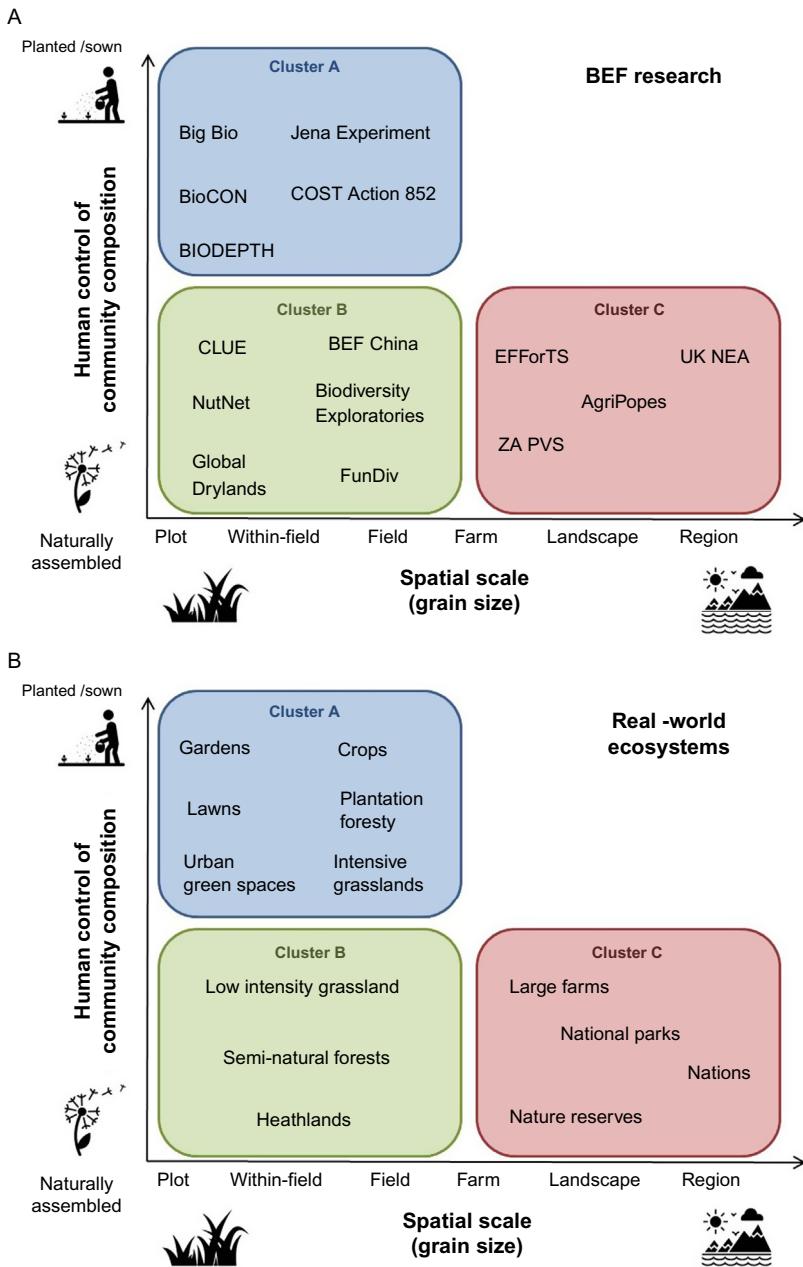
Fig. 1 Clusters of BEF research and their relation to real world ecosystems. (A) selected research projects, (B) selected 'real-world' ecosystems. Note that, as spatial scale increases, the user of research findings changes from individual local scale managers to governments and institutions and the form of transfer changes from management

experiments was to establish whether biodiversity could affect ecosystem functioning, and so they controlled for potentially confounding effects of environmental conditions, functional composition, individual density, and non-random assembly and disassembly processes (Schmid et al., 2002, Schmid and Hector, 2004, Eisenhauer et al., 2019, this issue). To achieve this, BEF experiments apply a diversity treatment, where varying levels of plants species richness are sown or planted, and ecosystem functioning is measured (Schmid et al., 2002: Bruelheide et al., 2014). As such studies are highly controlled (e.g. via randomized blocking, weeding and the homogenisation of growing conditions), diversity effects may be ascribed with confidence and detailed inferences can be made regarding the identity of the mechanisms driving biodiversity effects (Loreau and Hector, 2001).

While these experiments act as model systems for BEF research, with generally applicable results to a wide range of systems (Eisenhauer et al., 2016; Schmid and Hector, 2004), the direct application of these insights in the management of real-world ecosystems could be limited for several reasons. First, the sown or planted community (and its species richness) is maintained through the repeated removal of non-target species, which typically does not occur in real-world systems. As a result, communities may be present that would not persist without human intervention. Second, the species richness gradient tends to span levels of diversity (typically 1- <20 plant species) that are much lower than many semi-natural communities (Wilson et al., 2012). Third, the studies tend to be conducted in replicated plots smaller than $500\,m^2$ (Hector et al., 1999; Roscher et al., 2004; Tilman et al., 1996), with a median size of $3\,m^2$ (Cardinale et al., 2012). As such studies are labour-intensive, they also tend to be unreplicated at the landscape scale (but see Hector et al., 1999; Kirwan et al., 2007). However, the large number of experiments with comparable designs allows meta-level, large extent analyses to be conducted (Balvanera et al., 2006; Isbell et al., 2015b; Lefcheck et al., 2015; Verheyen et al., 2016; Craven et al., 2018).

Fig. 1—cont'd   practice recommendations to policy change, though these are clearly interrelated. Example references for the studies shown are: Jena experiment (Weisser et al., 2017), BigBio (Tilman et al., 2001), BioCON (Reich et al., 2001), COST Action 852 (Kirwan et al., 2007), BIODEPTH (Hector et al., 1999), BEF-China (Huang et al., 2018), CLUE (van der Putten et al., 2000), NutNet (Grace et al., 2016), Biodiversity Exploratories (Allan et al., 2015), Global Drylands (Maestre et al., 2012), FunDiv (van der Plas et al., 2016), EFForTS (Teuscher et al., 2016), AgriPopes (Emmerson et al., 2016), ZA PVS (Bretagnolle et al., 2018), UKNEA National Ecosystem Assessment (2011).

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

Research need and approach	Potential benefit to transfer	Examples or foundational 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) and 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) and 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) and 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) and Oliver et al. (2015)
Demonstrate the cost effectiveness of multispecies mixtures compared to existing management and develop technology that increases this (e.g. multicrop harvesters)	Unless clear benefits are demonstrated diversification may not be adopted	Finger and Buchmann (2015) and Blaauw and Isaacs (2014)
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) and 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)

Continued

Table 1 Research required to enable the real-world application of BEF research.—cont'd

Research need and approach	Potential benefit to transfer	Examples or foundational studies
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) and 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 and Knapp (2003) and 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 species are more likely to be lost	Soliveres et al. (2016a), Klein et al. (2003), and Lyons and Schwartz (2001)
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 (2019)
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) and 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) and 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)
Use upscaled measures to understand which taxa drive ecosystem services and disservices at landscape scales, and what factors drive the diversity of these taxa	Would allow important ecosystem service providers to be identified and managed appropriately	van der Plas et al. (2018), Winfree et al. (2018), and Grass et al. (2019)

Table 1 Research required to enable the real-world application of BEF research.—cont'd

Research need and approach	Potential benefit to transfer	Examples or foundational studies
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) and 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) (2019), European Landowners Organisation (ELO) (2019), F.R.A.N.Z. (2019), Conservation Evidence (2019) website, and RSPB Hope Farm (2019)
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	Geertsema et al. (2016) and Walter et al. (2017)

2.1 What can be transferred

BEF experiments were designed to provide general mechanistic insights into the BEF relationship. Nevertheless, the close control of plant community composition and their low species diversity means that findings from BEF experiments are potentially transferable to highly managed ecosystems, e.g., intensive agricultural grasslands, plantation forestry, gardens, sown communities found in urban green spaces or ecosystems restored from a heavily degraded state (Fig. 1B). Such systems tend to be managed intensively and at small scales, e.g., via the application of selective herbicides, weeding and fertilisation. As these systems typically contain fewer species than most semi-natural ecosystems, we predict that cluster A BEF research is best able to inform work related to diversification, rather than the impacts of species loss. BEF experiment results suggest that diversification of such systems would lead to considerable gains in the supply of some ecosystem services,

as numerous functions related to agricultural production and sustainability often increase with species diversity, including plant productivity, pollination, soil carbon storage and weed suppression (Isbell et al., 2017). Moreover, species–rich communities produce a more stable and constant yield (Craven et al., 2018; Isbell et al., 2015b), which may reduce risks to farmers (Finger and Buchmann, 2015).

Experimental results indicate that the benefits of diversification are greater when increasing diversity from low to intermediate levels (e.g. from 1 to 8 grassland species per m^2) than from medium to high (e.g. from 8 to 16), as the diversity-function relationship tends to saturate (Isbell et al., 2017). As species are typically grown in monocultures and in a wide range of low–diversity mixtures, data from these experiments can help to identify high performing species, but also high performing mixtures, for a range of ecosystem functions. Agronomists have conducted significant research on crop diversification for many years (Brooker et al., 2015; Vandermeer, 1992), and demonstrated that crop diversification can lead to various positive outcomes, such as increased primary crop yield and biocontrol (Iverson et al., 2014). Moreover, intercropping can improve yield stability (Raseduzzaman and Jensen, 2017), and more diverse mixtures of cover crops, especially those containing legumes, lead to multiple additional benefits (Blesh, 2018; Storkey et al., 2015), thus increasing their multifunctionality (defined here as ecosystem service multifunctionality, the co–supply of multiple ecosystem services relative to their human demand, Manning et al., 2018). Similarly, crop mixtures of multiple cultivars provide higher yields (Reiss and Drinkwater, 2018), and the mixing of rice varieties within a field reduces disease prevalence (Zhu et al., 2000). The frameworks and fundamental insights of BEF research may inform such research by identifying general rules governing complementary combinations of species and varieties (Brooker et al., 2015; Wright et al., 2017).

An additional benefit of BEF experiments is that they often provide information on a wider range of ecosystem services than many agricultural experiments and agronomic analyses, which tend to focus on yield and its sustainability, e.g., weed control and nutrient cycling (Meyer et al., 2018). Mixtures that promote the supply of multiple ecosystem services simultaneously may therefore be identified from BEF studies (Baeten et al., 2019; Storkey et al., 2015). Further evidence of existing BEF transfer comes from grassland studies, which indicate that there are multiple benefits of diversifying agroecosystems in terms of grass yield and reduced weed

abundance (Finn et al., 2013). Studies have also shown that diverse grassland mixtures produce greater bioenergy yields (Khalsa et al., 2014; Tilman et al., 2006). However, another study of bioenergy production in grass mixtures showed that diverse mixtures were not more productive than currently used monocultures, thus showing that diversification might not always promote bioenergy production (Dickson and Gross, 2015). Even in the absence of positive impacts of diversity on productivity, other benefits may be realized; diverse bioenergy landscapes can promote the supply of other ecosystem services including greenhouse gas mitigation, pest suppression, pollination, and bird watching potential (Werling et al., 2014).

A number of other avenues of experimental BEF research have the capacity to inform the management of intensive systems. BEF experiments show that damage to plant growth and productivity from plant pathogens and pests is often weaker in more diverse communities, both aboveground (Civitello et al., 2015; Otway et al., 2005) and belowground (Maron et al., 2011; Schnitzer et al., 2011). Accordingly, information from BEF experiments on plant-soil feedbacks (e.g. Vogel et al., 2019 this issue) could potentially help to devise effective crop rotation sequences, e.g., by identifying consistent antagonistic or synergistic feedbacks between functional groups when grown together or in sequence (Barel et al., 2018; Ingerslew and Kaplan, 2018). The insights of BEF experiments are also applicable to gardens and green roof planting (Lundholm et al., 2010) and the restoration of highly degraded ecosystems. Here it may be possible to determine species mixtures or particular functional trait combinations, which, when sown or planted, deliver desired functions, such as soil aggregate stability and soil organic matter accumulation (Gould et al., 2016; Kollmann et al., 2016; Lange et al., 2015; Yang et al., 2019). In restoration, another promising approach would be to identify and sow mixtures of species that facilitate each other as this is a key mechanism underlying biodiversity effects in harsh environments (Wright et al., 2017). Finally, evidence from forests suggests that similar or higher amounts of timber production can be achieved in mixed plantations of native species compared to monocultures of plantation species, and that co-benefits, e.g., to biodiversity conservation, would also be realized (Gamfeldt et al., 2013; Huang et al., 2018; Hulvey et al., 2013; Pretzsch and Schütze, 2009). As with crops, the results of BEF studies can also be used to indicate the tree species mixtures that best achieve this multifunctionality (Baeten et al., 2019; Teuscher et al., 2016).

2.2 Barriers to transfer and directions for future research

While the plant communities of BEF experiments and human–dominated ecosystems share similarities, there are also marked differences. For instance, the species composition in BEF experiments is randomly assembled and they are usually performed in unfertilized, pesticide-free, unirrigated systems. In contrast, in intensively managed real–world systems, prior knowledge has led managers to select high performing, but often low diversity, mixtures by sowing and planting species that deliver high levels of desired services, and/or encouraging these via pesticide application, irrigation and fertilisation. The benefits of diversification therefore need to be demonstrated relative to these intensive low diversity communities, rather than the random low diversity assemblages found in BEF experiments. For example, in European grasslands farmers typically sow or maintain mixtures of a single grass, *Lolium perenne,* and a single legume, *Trifolium repens,* to which fertilizers are also applied (Peeters et al., 2014). Such a mixture clearly differs from the random species-poor mixtures of grassland biodiversity experiments. It is unclear if the relatively diverse and high-functioning communities of biodiversity experiments are generally able to deliver yield of a similar or higher quality, quantity and reliability. However, it has been demonstrated that diversification from 1–2 to 3–4 species provides significant increases in grassland yield and higher resistance to weed invasion (Finn et al., 2013; Kirwan et al., 2007; Nyfeler et al., 2009). We hypothesize that the species-poor communities found in intensively managed systems are more likely to resemble the high performing species–poor communities of BEF experiments (e.g. those dominated by tall grasses of fertile conditions) than the low performing communities, which may struggle to persist without regular weeding and close control (e.g. those containing only a few small herbs). In contrast, the low diversity situations found in experiments, where potentially dominant species are missing, could be relevant to isolated habitat patches, where species cannot disperse to potentially suitable conditions and the species pool is restricted.

As described above, current research suggests that links between BEF and agronomic research are beginning to emerge. However, current studies do not cover the wide range of situations in which diversification could be beneficial to agroecosystems. To the best of our knowledge, little work has yet made the transition to widespread adoption, an exception being the standard mixtures for forage production in Switzerland (see Fig. 2 for details), This lack of adoption highlights knowledge exchange as an important bottleneck

Fig. 2 Swiss grassland diversification. In Switzerland species rich semi-natural grasslands (left) can decline to a more species-poor state (right) if fertilized and mown frequently. To counteract this loss many species rich sites are maintained via agri-environment policy schemes (Kampmann et al., 2012) and Swiss researchers have developed diversified seed mixtures suitable for a wide range of conditions that have been adopted by many Swiss farmers (Suter et al., 2017). We postulate that this adoption is likely to be attributable to a range of factors including: a strong cultural valuation of grassland, a clear mandate of agriculture to manage sustainably (in Swiss Constitution, article 104), generous agri-environment compensation schemes for many grassland types, and a strong focus on applied grassland research that has investigated which mixtures work over different time horizons (e.g. annual to permanent) and environmental conditions (moisture and elevational gradients) (e.g. Suter et al., 2015). Finally, there is effective communication from both researchers (e.g. Agroscope) and the Swiss grassland society (AGFF, 2019), which contains many farmers as members. Future BEF transfer work could investigate the role of such factors in successful transfer. Photo credits Peter Manning.

and another future need. To enable this, future BEF experiments could increase their relevance for management by drawing experimental communities from species pools that contain potentially useful and manageable species, and performing experiments in settings that are similar to those found in land use systems (e.g. fertilized or grazed grasslands). In this way, communities that are manageable and multifunctional may also be identified, and specific mixtures can be recommended (e.g. current policy in Switzerland). These should be cost-efficient and self-supporting and thus easily adapted and maintained by land managers.

Results on the relationship between biodiversity and the stability of ecosystem functions and services also require re-interpretation if they are to inform ecosystem management. While definitions of stability very greatly (Grimm and Wissel, 1997), BEF studies typically measure stability as the coefficient of variation (e.g. Craven et al., 2018; Knapp and van der Heijden, 2018), the resistance to perturbations, or the rate of recovery following these (Isbell et al., 2015b). In contrast, ecosystem managers often

perceive stability differently (Donohue et al., 2016); while reliability is appreciated, and there are minimum levels of ecosystem service supply that are acceptable and over-performance (e.g. high productivity in favourable weather years, Wright et al., 2015) is often appreciated. Therefore, alternative measures of stability, e.g., that measure the number of years in which the supply of services exceed an acceptable threshold (Oliver et al., 2015), need to be employed if diversity-stability relationships are to be determined meaningfully for agroecosystems.

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

research involving economic and/or multiple stakeholder-based assessments of the value of the diverse systems relative to current and future systems and practices (Jackson et al., 2012; Geertsema et al., 2016; Bretagnolle et al., 2018; Kleijn et al., 2018) (Table 1).

3. Small-grain studies with low experimental control (Cluster B)

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

The design of studies in this cluster limits interpretation about the cause of biodiversity effects as data for monoculture performances are usually unavailable, meaning that the mechanisms underlying biodiversity effects cannot be estimated (Loreau and Hector, 2001). This is unfortunate as these processes may differ in their strength compared to biodiversity experiments. For example, in mature communities, species may show higher levels of niche differentiation at both between and within species levels (Guimarães-Steinicke et al., 2019, this issue; Zuppinger-Dingley et al., 2014). A final property differentiating cluster B studies from those of cluster A is that variation in the diversity of other trophic levels is a complex product of responses to environmental drivers and concurrent changes in all trophic levels (Soliveres et al., 2016a, 2016b; Tscharntke et al., 2005), rather than primarily driven by variation in the diversity of primary producers (Scherber et al., 2010).

3.1 What can be transferred

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

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

3.2 Barriers to transfer and directions for future research

For research in cluster B to become more directly transferable to the management of semi-natural ecosystems, greater confidence in the mechanisms underlying real-world BEF relationships is needed. While management recommendations may be drawn from selected case studies such as those

presented above, a general understanding of the relative and interacting roles of environmental covariates, direct effects of global change drivers and various facets of diversity and compositional change is lacking (van der Plas, 2019). Biodiversity could play an important role in maintaining ecosystem function in real world ecosystems. Yet, whether loss of a few species at this scale makes a strong contribution to function, relative to these other drivers, has been only been tested in a limited number of cases (e.g. Allan et al., 2015; Grace et al., 2016; Manning et al., 2006; Winfree et al., 2015), and inconsistently, making generalisation difficult (van der Plas, 2019). To address this issue, observational studies need to ensure that factors such as abundance and functional composition are properly controlled for statistically. Predictions of the impacts of drivers on ecosystem services can be made by combining (a) estimates of expected biodiversity change according to different global change drivers across a range of conditions (e.g. Bjorkman et al., 2018; Grace et al., 2016; Hautier et al., 2018), (b) knowledge of how great a difference to functions and services such changes will make (e.g. Craven et al., 2018), and (c) ecosystem service production functions (Isbell et al., 2015a). This in turn allows for estimates of where ecosystem service-based arguments for conservation are strongest. Such predictions, if verified, could then form a sound basis for management decisions.

Transfer would also be enabled by a new generation of experiments. These could include a wider range of non-random extinction scenarios, assessments of the relative importance of abiotic drivers of function and biodiversity (e.g. Isbell et al., 2013; Manning et al., 2006), and the reduction of diversity from high to intermediate levels (Zobel et al., 1994), in order to verify, or refute the results of observational studies. To do this, manipulations such as the manipulation of dominance and functional composition, trait dissimilarity, or other aspects of biodiversity could be employed (Cross and Harte, 2007; Manning et al., 2006; Smith and Knapp, 2003). Manipulations that simulate the homogenisation of biota (i.e. the loss of beta diversity, while alpha diversity remains unchanged), may also prove informative, as this may be as, or more, common than alpha diversity loss in real-world ecosystems (Flohre et al., 2011; Vellend et al., 2013; Dornelas et al., 2014; Gossner et al., 2016; Wardle, 2016). Finally, it may be possible to link community assembly mechanisms (e.g. founder effects and habitat filtering) and functional BEF research to identify how to increase species richness and promote certain ecosystem functions, information that would be particularly useful in ecosystem restoration (Bullock et al., 2007; Kirmer et al., 2012; Stein et al., 2008; Weidlich et al., 2018) (Table 1).

Work is also needed in converting the measures of ecosystem function commonly taken in ecological studies into measures of ecosystem services that are of relevance to stakeholders (Mace et al., 2012; Kleijn et al., 2018). This requires the development of new metrics, e.g., trait measures that link to nutritional quality or cultural services such as aesthetic appeal. Applied studies could explicitly measure relevant ecosystem services, e.g., by involving stakeholders, assessing which services are most important to them, and adapting function measures to quantify these (King et al., 2015; Manning et al., 2018; Martín–López et al., 2012). This approach, and many of the others outlined above requires inter- and transdisciplinary research involving stakeholders and researchers from other disciplines, e.g., with farmers, local governments, agronomists and economists.
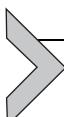
4. Large-grain studies without experimental control (Cluster C)

The third cluster (C) contains BEF studies that cover large areas (from 100 m^2 to landscapes) (e.g. Garibaldi et al., 2013; Larsen et al., 2005; Winfree et al., 2018). Due to the huge efforts required to manipulate diversity at a large spatial and temporal grain (Teuscher et al., 2016), such studies tend to be observational, comparative, and of low replication, although the large number of such studies has allowed for meta-level analyses to be conducted (Lichtenberg et al., 2017). The focal study organisms also tend to be invertebrates, particularly pollinators, instead of plants. The measurement of biodiversity (e.g. species richness and functional diversity) is also often limited in these studies due to the effort required to measure it directly over large areas. As a result, it is often landscape variables, such as landscape configuration and the proportion of different land uses that are related to function, rather than diversity (e.g. Bosem Baillod et al., 2017; Hass et al., 2018). These landscape properties may influence the dispersal, abundance and diversity of organisms within the landscape, and may also correlate with management factors and abiotic drivers of ecosystem function (Dominik et al., 2018; Gámez–Virués et al., 2015; Lindborg et al., 2017). As a result of these covariances, the role of biodiversity in driving ecosystem functioning cannot always be confidently ascribed (Tscharntke et al., 2016).

Within this cluster, we also place remote sensing studies (e.g. Oehri et al., 2017) and national and regional correlational studies (e.g. Anderson et al., 2009). In these, biodiversity can only be measured using proxies or with presence/absence data within large grid cells (e.g. 10×10 km), e.g., from

national monitoring schemes. These coarse biodiversity measures are then correlated with ecosystem service proxy measures such as carbon storage and recreational use. These studies often lack a strong mechanistic basis, and focus instead on how biodiversity co-varies with ecosystem services (e.g. Anderson et al., 2009; Maskell et al., 2013). Even where covariates are included and mechanistic relationships postulated (e.g. Oehri et al., 2017; Duffy et al., 2017), causal links are hard to infer due to the strong covariance between biodiversity and other drivers, and the high probability of missing, or improperly measuring, important covariates.

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

4.1 What can be transferred

As the studies of cluster C are performed in real landscapes, and as management is often conducted at large scales (e.g. by farmers or foresters), research findings from this cluster are potentially of high relevance to policy and large-scale management, e.g., via payments for ecosystem service schemes. In recent years, a number of studies have demonstrated large-scale benefits of landscapes with high diversity of crops and non-crop habitats, which support higher biodiversity (Gardiner et al., 2009; Redlich et al., 2018). These benefits include more effective pollination and biological pest control (Garibaldi et al., 2013; Winfree et al., 2018). By showing how diversity and diversification practices influence ecosystem service delivery, these practices can then be incorporated into agronomic considerations (Rosa-Schleich et al., 2019) and into agri-environment policy (Garibaldi et al., 2014). Studies at this scale also complement those of the other clusters by showing that biodiversity not only promotes ecosystem function and services at the plot scale but also via spillover effects into the surrounding landscape, with ecosystem service benefits including pest suppression, pollination, and bird watching potential (Blitzer et al., 2012; Werling et al., 2014). However, biodiversity does not always promote function at these scales. For example, natural enemy diversity does not

always relate to pest abundance, nor higher crop yields (Tscharntke et al., 2016), and in some cases biodiversity does not control pests as effectively as pesticides (Samnegard et al., 2019).

4.2 Barriers to transfer and directions for future research

The observational nature of most research in this cluster means that the exact role of diversity in driving ecosystem function and providing ecosystem services at these scales is hard to ascertain. This general limitation is compounded by several other barriers which can prevent transfer to landscape management and policy. First, several processes could drive BEF relationships at landscape scales that do not operate at the smaller grain size of clusters A and B, and as a result are little acknowledged in BEF research, outside of theory (Lindborg et al., 2017; Loreau et al., 2003; Tscharntke et al., 2012). These include the spatial processes that maintain diversity, the matching between species and environmental conditions in which they perform well (Leibold et al., 2017; Mori et al., 2018), and the potential for different species to provide different functions and services in different patches of the landscape, thus boosting landscape multifunctionality (van der Plas et al., 2016; van der Plas et al., 2019). The strength and role of such mechanisms clearly needs to be demonstrated. Another key problem in transferring BEF research to large scales is that landscape managers typically seek to simultaneously promote multiple ecosystem services, i.e., the multifunctionality of landscapes, not single ecosystem functions at the plot scale (Kremen and Merenlender, 2018; Manning et al., 2018). A focus on single functions is problematic if they trade-off and the components of diversity that boost some ecosystem services diminish others. For example, the maintenance of biodiversity-rich habitats may add resilience to multiple ecosystem functions at the landscape scale, but also occupies land that could be used for crop production.

New research approaches are required to overcome the difficulties in identifying how biodiversity controls ecosystem functioning at large scales, and how biodiversity may be conserved and promoted to increase the supply of ecosystem services. First, to ensure that service measures are of relevance to stakeholders, we require a better understanding of which services are demanded by different stakeholders, and at which different temporal and spatial scales, so that relevant indicator variables or ecosystem service production functions can be used (Tallis, 2011). A more holistic approach,

which accounts for the relative demand for different ecosystem services and how this changes with socio-economic context, is therefore required, e.g., to assess how much land can be returned to a high biodiversity condition while maintaining desired levels of food production and other ecosystem services (Clough et al., 2011; Kremen and Merenlender, 2018; Manning et al., 2018). Such studies should also identify what drives patterns of land use and management and hence biodiversity loss, so that appropriate interventions can be identified (Grass et al., 2019).

To consider landscape multifunctionality and its dependence on biodiversity, multiple ecosystem services need to be scaled up in space and time, which is challenging. Some of the functions that can be measured at the plot scale can be 'linearly' scaled up, e.g., by using remote sensing proxies of diversity and functional traits, and interpolated maps, e.g., of climate and soil properties (Manning et al., 2015; van der Plas et al., 2018). Others, however, require an understanding of spatial interactions that makes their upscaling more complex, e.g., pollination and nutrient leaching (Koh et al., 2016; Lindborg et al., 2017). Furthermore, some services that operate at large scales (e.g. flood control, landscape aesthetics) cannot be predicted and scaled up from small-scale measures. Therefore, new procedures and methods are needed to quantify large-scale multifunctionality and the role of biodiversity in driving it. There have been calls for landscape-scale experiments to address these issues (Koh et al., 2009; Landis, 2017). One example is the recent EFForTS project in which "tree islands" of varying size and tree diversity (0–6 species) have been planted in oil-palm clearings (Teuscher et al., 2016). Initial results indicate no economic trade-off: the islands generate yield gains which compensate for the reduced number of oil palms (Gérard et al., 2017). However, the high financial cost and/or logistical effort of such experiments means it may be more realistic to use biophysical models in most cases. Unfortunately, such models do not currently fully represent the complexity of biodiversity or its relationship with ecosystem functions and services (Lavorel et al., 2017).

To understand biodiversity-landscape multifunctionality relationships, a greater knowledge of which aspects of diversity underpin different ecosystem services is also required. While knowledge exists regarding the drivers of many ecosystem service provider groups at the landscape scale (e.g. plants, birds, butterflies and pollinators, Roschewitz et al., 2005; Rösch et al., 2015; Kormann et al., 2015; Grab et al., 2019), this understanding needs to be extended to other groups, including soil microbes and soil fauna. Similarly,

understanding of how spatial biodiversity dynamics affect functions and the services they underpin needs to be extended to taxa involved in services other than pest control and pollination (Table 1). In some cases, there may be trade-offs between services, e.g., if the conditions that maximize the diversity of one taxa do not favour another (van der Plas et al., 2019). This research may also demonstrate that when it comes to real-world eco-system services and landscape-level multifunctionality, biodiversity effects are not easily generalizable, but depend on the context. Thus, the rules of this context-dependency need to be identified (Allan et al., 2015; Birkhofer et al., 2018; Samnegard et al., 2019). Doing this will limit uncer-tainty; managers could be less reluctant to manage for biodiversity when the degree to which it provides ecosystem service benefits at larger scales has been clearly demonstrated. In semi-natural ecosystems the promotion of the biodiversity components underpinning ecosystem services are most likely to be achieved via management options that are simple and effective over large areas, and so the practices that would promote the desired facets of biodiversity, e.g., mowing or the introduction of selective grazers, may need to be identified.
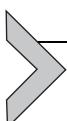
5. Conclusion

A vast array of BEF studies has taught us much about the complex rela-tionship between biodiversity and ecosystem functioning. In this article, we argue that with some re-analysis and re-interpretation, some of this research could be directly transferred to policy and management, where practitioners could use its insights to guide the diversification of agricultural and other human-dominated ecosystems, and inform the conservation of biodiversity in semi-natural ecosystems. However, there are numerous challenges to the transfer of BEF research to more applied research and practice, and we argue that these challenges differ depending on the spatial grain of the study and the degree of community manipulation. While acknowledging the differences in transferability between these clusters of BEF research may help resolve the ongoing debate about relevance of BEF findings a new generation of BEF research is also required. This would involve the merging and con-necting research between the current clusters, e.g., the setup of a new gen-eration of biodiversity experiments that bridge the gap between current BEF experiments and observational studies. These should be complemented by new observational studies which more comprehensively account for

covarying factors and which better acknowledge the link between ecosystem function and ecosystem services (Table 1).

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

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

Acknowledgements

References

Arbeitsgemeinschaft zur Förderung des Futterbaues (AGFF), 2019. http://www.agff.ch/deutsch/aktuell.html.

Allan, E., Manning, P., Alt, F., Binkenstein, J., Blaser, S., Blüthgen, N., Böhm, S., Grassein, F., Hölzel, N., Klaus, V.H., Kleinebecker, T., 2015. Land use intensification alters ecosystem multifunctionality via loss of biodiversity and changes to functional composition. Ecol. Lett. 18, 834–843.

Anderson, B.J., Armsworth, P.R., Eigenbrod, F., Thomas, C.D., Gillings, S., Heinemeyer, A., Roy, D.B., Gaston, K.J., 2009. Spatial covariance between biodiversity and other ecosystem service priorities. J. Appl. Ecol. 46, 888–896.

Baeten, L., Bruelheide, H., van der Plas, F., Kambach, S., Ratcliffe, S., Jucker, T., Allan, E., Ampoorter, E., Barbaro, L., Bastias, C.C., Bauhus, J., 2019. Identifying the tree species compositions that maximize ecosystem functioning in European forests. J. Appl. Ecol. 56, 733–744.

Balvanera, P., Pfisterer, A.B., Buchmann, N., He, J.S., Nakashizuka, T., Raffaelli, D., Schmid, B., 2006. Quantifying the evidence for biodiversity effects on ecosystem functioning and services. Ecol. Lett. 9, 1146–1156.

Barel, J.M., Kuyper, T.W., de Boer, W., Douma, J.C., De Deyn, G.B., 2018. Legacy effects of diversity in space and time driven by winter cover crop biomass and nitrogen concentration. J. Appl. Ecol. 55, 299–310.

Barnes, A.D., Weigelt, P., Jochum, M., Ott, D., Hodapp, D., Haneda, N.F., Brose, U., 2016. Species richness and biomass explain spatial turnover in ecosystem functioning across tropical and temperate ecosystems. Philos. Trans. R. Soc., B 371, 20150279.

Binder, S., Isbell, F., Polasky, S., Catford, J.A., Tilman, D., 2018. Grassland biodiversity can pay. Proc. Natl. Acad. Sci. 115, 3876–3881.

Birkhofer, K., Andersson, G.K., Bengtsson, J., Bommarco, R., Dänhardt, J., Ekbom, B., Ekroos, J., Hahn, T., Hedlund, K., Jönsson, A.M., Lindborg, R., 2018. Relationships between multiple biodiversity components and ecosystem services along a landscape complexity gradient. Biol. Conserv. 218, 247–253.

Bjorkman, A.D., Myers-Smith, I.H., Elmendorf, S.C., Normand, S., Rüger, N., Beck, P.S., Blach-Overgaard, A., Blok, D., Cornelissen, J.H.C., Forbes, B.C., Georges, D., et al., 2018. Plant functional trait change across a warming tundra biome. Nature 562, 57.

Blaauw, B.R., Isaacs, R., 2014. Flower plantings increase wild bee abundance and the pollination services provided to a pollination-dependent crop. J. Appl. Ecol. 51, 890–898.

Blesh, J., 2018. Functional traits in cover crop mixtures: biological nitrogen fixation and multifunctionality. J. Appl. Ecol. 55, 38–48.

Blitzer, E.J., Dormann, C.F., Holzschuh, A., Klein, A.M., Rand, T.A., Tscharntke, T., 2012. Spillover of functionally important organisms between managed and natural habitats. Agr. Ecosyst. Environ. 146, 34–43.

Bosem Baillod, A., Tscharntke, T., Clough, Y., Batáry, P., 2017. Landscape-scale interactions of spatial and temporal cropland heterogeneity drive biological control of cereal aphids. J. Appl. Ecol. 54, 1804–1813.

Bretagnolle, V., Berthet, E., Gross, N., Gauffre, B., Plumejeaud, C., Houte, S., Badenhausser, I., Monceau, K., Allier, F., Monestiez, P., Gaba, S., 2018. Towards sustainable and multifunctional agriculture in farmland landscapes: lessons from the integrative approach of a French LTSER platform. Sci. Total Environ. 627, 822–834.

Brooker, R.W., Bennett, A.E., Cong, W.F., Daniell, T.J., George, T.S., Hallett, P.D., Hawes, C., Iannetta, P.P., Jones, H.G., Karley, A.J., Li, L., 2015. Improving intercropping: a synthesis of research in agronomy, plant physiology and ecology. New Phytol. 206, 107–117.

Bruelheide, H., Nadrowski, K., Assmann, T., Bauhus, J., Both, S., Buscot, F., Chen, X.Y., Ding, B., Durka, W., Erfmeier, A., Gutknecht, J.L., 2014. Designing forest biodiversity

experiments: general considerations illustrated by a new large experiment in subtropical China. Methods Ecol. Evol. 5, 74–89.

Bullock, J.M., Pywell, R.F., Walker, K.J., 2007. Long-term enhancement of agricultural production by restoration of biodiversity. J. Appl. Ecol. 44, 6–12.

Cardinale, B.J., Duffy, J.E., Gonzalez, A., Hooper, D.U., Perrings, C., Venail, P., Narwani, A., Mace, G.M., Tilman, D., Wardle, D.A., Kinzig, A.P., 2012. Biodiversity loss and its impact on humanity. Nature 486, 59.

Civitello, D.J., Cohen, J., Fatima, H., Halstead, N.T., Liriano, J., McMahon, T.A., Ortega, C.N., Sauer, E.L., Sehgal, T., Young, S., Rohr, J.R., 2015. Biodiversity inhibits parasites: broad evidence for the dilution effect. Proc. Natl. Acad. Sci. 112, 8667–8671.

Clark, C.M., Tilman, D., 2010. Recovery of plant diversity following N cessation: effects of recruitment, litter, and elevated N cycling. Ecology 91, 3620–3630.

Clough, Y., Barkmann, J., Juhrbandt, J., Kessler, M., Wanger, T.C., Anshary, A., Buchori, D., Cicuzza, D., Darras, K., Putra, D.D., Erasmi, S., et al., 2011. Combining high biodiversity with high yields in tropical agroforests. Proc. Natl. Acad. Sci. 108, 8311–8316.

Clough, Y., Krishna, V.V., Corre, M.D., Darras, K., Denmead, L.H., Meijide, A., Moser, S., Musshoff, O., Steinebach, S., Veldkamp, E., Allen, K., et al., 2016. Land-use choices follow profitability at the expense of ecological functions in Indonesian smallholder landscapes. Nat. Commun. 7, 13137.

Conservation Evidence, 2019. https://www.conservationevidence.com/.

Craven, D., Eisenhauer, N., Pearse, W.D., Hautier, Y., Roscher, C., Isbell, F., Bahn, M., Beierkuhnlein, C., Bönisch, G., Buchmann, N., Byun, C., et al., 2018. Multiple facets of biodiversity drive the diversity-stability relationship. Nat Ecol Evol 2, 1.

Cross, M.S., Harte, J., 2007. Compensatory responses to loss of warming-sensitive plant species. Ecology 88, 740–748.

Dıaz, S., Symstad, A.J., Chapin III, F.S., Wardle, D.A., Huenneke, L.F., 2003. Functional diversity revealed by removal experiments. Trends Ecol. Evol. 18, 140–146.

Díaz, S., Demissew, S., Carabias, J., Joly, C., Lonsdale, M., Ash, N., Larigauderie, A., Adhikari, J.R., Arico, S., Báldi, A., Bartuska, A., et al., 2015. The IPBES conceptual framework—connecting nature and people. Curr. Opin. Environ. Sustain. 14, 1–16.

Díaz, S., Pascual, U., Stenseke, M., Martín-López, B., Watson, R.T., Molnár, Z., Hill, R., Chan, K.M., Baste, I.A., Brauman, K.A., Polasky, S., 2018. Assessing nature's contributions to people. Science 359, 270–272.

Dickson, T.L., Gross, K.L., 2015. Can the results of biodiversity-ecosystem productivity studies be translated to bioenergy production? PLoS One 10, e0135253.

Dominik, C., Seppelt, R., Horgan, F.G., Settele, J., Václavík, T., 2018. Landscape composition, configuration, and trophic interactions shape arthropod communities in rice agroecosystems. J. Appl. Ecol. 55, 2461–2472.

Donohue, I., Hillebrand, H., Montoya, J.M., Petchey, O.L., Pimm, S.L., Fowler, M.S., Healy, K., Jackson, A.L., Lurgi, M., McClean, D., O'Connor, N.E., O'Gorman, E.J., Yang, Q., Adler, F., 2016. Navigating the complexity of ecological stability. Ecol. Lett. 19, 1172–1185.

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

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

Eisenhauer, N., Barnes, A.D., Cesarz, S., Craven, D., Ferlian, O., Gottschall, F., Hines, J., Sendek, A., Siebert, J., Thakur, M.P., Türke, M., 2016. Biodiversity–ecosystem function experiments reveal the mechanisms underlying the consequences of biodiversity change in real world ecosystems. J. Veg. Sci. 27, 1061–1070.

Eisenhauer, N., Schielzeth, H., Barnes, A.D., Barry, K.E., Bonn, A., Brose, U., Bruelheide, H., Buchmann, N., Buscot, F., Ebeling, A., et al., 2019. A multitrophic perspective on biodiversity–ecosystem functioning research. Adv. Ecol. Res. this issue.

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

European Landowners Organisation (ELO), 2019. https://www.europeanlandowners.org/.

Fanin, N., Gundale, M.J., Farrell, M., Ciobanu, M., Baldock, J.A., Nilsson, M.C., Kardol, P., Wardle, D.A., 2018. Consistent effects of biodiversity loss on multifunctionality across contrasting ecosystems. Nat. Ecol. Evol. 2, 269.

Fazey, I., Evely, A.C., Reed, M.S., Stringer, L.C., Kruijsen, J., White, P.C., Newsham, A., Jin, L., Cortazzi, M., Phillipson, J., Blackstock, K., 2013. Knowledge exchange: a review and research agenda for environmental management. Environ. Conserv. 40, 19–36.

Finger, R., Buchmann, N., 2015. An ecological economic assessment of risk-reducing effects of species diversity in managed grasslands. Ecol. Econ. 110, 89–97.

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

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

Forum for the Future of Agriculture (FFA), 2019. http://www.forumforagriculture.com/.

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

Für Ressourcen, Agrarwirtschaft & Naturschutz mit Zukunft (F.R.A.N.Z.), 2019, F.R.A.N.Z. www.franz-projekt.de.

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

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M.C., Fröberg, M., Stendahl, J., Philipson, C.D., Mikusiński, G., 2013. Higher levels of multiple ecosystem services are found in forests with more tree species. Nat. Commun. 4, 1340.

Gardiner, M.M., Landis, D.A., Gratton, C., DiFonzo, C.D., O'neal, M., Chacon, J.M., Wayo, M.T., Schmidt, N.P., Mueller, E.E., Heimpel, G.E., 2009. Landscape diversity enhances biological control of an introduced crop pest in the north-Central USA. Ecol. Appl. 19, 143–154.

Garibaldi, L.A., Steffan-Dewenter, I., Winfree, R., Aizen, M.A., Bommarco, R., Cunningham, S.A., Kremen, C., Carvalheiro, L.G., Harder, L.D., Afik, O., Bartomeus, I., et al., 2013. Wild pollinators enhance fruit set of crops regardless of honey bee abundance. Science 339, 1608–1611.

Garibaldi, L.A., Carvalheiro, L.G., Leonhardt, S.D., Aizen, M.A., Blaauw, B.R., Isaacs, R., Kuhlmann, M., Kleijn, D., Klein, A.M., Kremen, C., Morandin, L., et al., 2014. From research to action: enhancing crop yield through wild pollinators. Front. Ecol. Environ. 12, 439–447.

Garibaldi, L.A., Bartomeus, I., Bommarco, R., Klein, A.M., Cunningham, S.A., Aizen, M.A., Boreux, V., Garratt, M.P., Carvalheiro, L.G., Kremen, C., Morales, C.L., 2015.

Trait matching of flower visitors and crops predicts fruit set better than trait diversity. J. Appl. Ecol. 52, 1436–1444.

Gaujour, E., Amiaud, B., Mignolet, C., Plantureux, S., 2012. Factors and processes affecting plant biodiversity in permanent grasslands. A review. Agron. Sustain. Dev. 32, 133–160.

Geertsema, W., Rossing, W.A., Landis, D.A., Bianchi, F.J., Van Rijn, P.C., Schaminée, J.H., Tscharntke, T., Van Der Werf, W., 2016. Actionable knowledge for ecological intensification of agriculture. Front. Ecol. Environ. 14, 209–216.

Gérard, A., Wollni, M., Hölscher, D., Irawan, B., Sundawati, L., Teuscher, M., Kreft, H., 2017. Oil-palm yields in diversified plantations: initial results from a biodiversity enrichment experiment in Sumatra, Indonesia. Agr. Ecosyst. Environ. 240, 253–260.

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

Gould, I.J., Quinton, J.N., Weigelt, A., De Deyn, G.B., Bardgett, R.D., 2016. Plant diversity and root traits benefit physical properties key to soil function in grasslands. Ecol. Lett. 19, 1140–1149.

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

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

Grass, I., Loos, J., Baensch, S., Batáry, P., Librán-Embid, F., Ficiciyan, A., Klaus, F., Riechers, M., Rosa, J., Tiede, J., Udy, K., 2019. Land-sharing/−sparing connectivity landscapes for ecosystem services and biodiversity conservation. People and Nature 1, 262–272.

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

Guimarães-Steinicke, C., Weigelt, A., Ebeling, A., Eisenhauer, N., Duque-Lazo, J., Reu, B., Roscher, C., Wagg, C., Wirth, C., 2019. Terrestrial laser scanning reveals temporal changes in biodiversity mechanisms driving grassland productivity. Adv. Ecol. Res. this issue.

Hass, A.L., Kormann, U.G., Tscharntke, T., Clough, Y., Baillod, A.B., Sirami, C., Fahrig, L., Martin, J.L., Baudry, J., Bertrand, C., Bosch, J., 2018. Landscape configurational heterogeneity by small-scale agriculture, not crop diversity, maintains pollinators and plant reproduction in western Europe. Proc. R. Soc. B Biol. Sci. 285, 20172242.

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

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

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

Hines, J., Ebeling, A., Barnes, A., Brose, U., Scherber, C., Scheu, S., Tscharntke, T., Weisser, W.W., Giling, D.P., Klein, A., Eisenhauer, N., et al., 2019. Mapping change in biodiversity and ecosystem function research: food webs foster integration of experiments and science policy. Adv. Ecol. Res. this issue.

Hooper, D.U., Chapin, F., Ewel, J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J., Lodge, D., Loreau, M., Naeem, S., et al., 2005. Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. Ecol. Monogr. 75, 3–35.

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

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

Hulvey, K.B., Hobbs, R.J., Standish, R.J., Lindenmayer, D.B., Lach, L., Perring, M.P., 2013. Benefits of tree mixes in carbon plantings. Nat. Clim. Chang. 3, 869.

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

Ingerslew, K.S., Kaplan, I., 2018. Distantly related crops are not better rotation partners for tomato. J. Appl. Ecol. 55, 2506–2516.

Isbell, F., Reich, P.B., Tilman, D., Hobbie, S.E., Polasky, S., Binder, S., 2013. Nutrient enrichment, biodiversity loss, and consequent declines in ecosystem productivity. Proc. Natl. Acad. Sci. 110, 11911–11916.

Isbell, F., Tilman, D., Polasky, S., Loreau, M., 2015a. The biodiversity-dependent ecosystem service debt. Ecol. Lett. 18, 119–134.

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

Isbell, F., Adler, P.R., Eisenhauer, N., Fornara, D., Kimmel, K., Kremen, C., Letourneau, D.K., Liebman, M., Polley, H.W., Quijas, S., 2017. Benefits of increasing plant diversity in sustainable agroecosystems. J. Ecol. 105, 871–879.

Iverson, A.L., Marín, L.E., Ennis, K.K., Gonthier, D.J., Connor-Barrie, B.T., Remfert, J.L., Cardinale, B.J., Perfecto, I., 2014. Do polycultures promote win-wins or trade-offs in agricultural ecosystem services? A meta-analysis. J. Appl. Ecol. 51, 1593–1602.

Jackson, L.E., Pulleman, M.M., Brussaard, L., Bawa, K.S., Brown, G.G., Cardoso, I.M., De Ruiter, P.C., García-Barrios, L., Hollander, A.D., Lavelle, P., Ouédraogo, E., 2012. Social–ecological and regional adaptation of agrobiodiversity management across a global set of research regions. Glob. Environ. Chang. 22, 623–639.

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

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

King, E., Cavender-Bares, J., Balvanera, P., Mwampamba, T., Polasky, S., 2015. Trade-offs in ecosystem services and varying stakeholder preferences: evaluating conflicts, obstacles, and opportunities. Ecol. Soc. 20, 25.

Kirmer, A., Baasch, A., Tischew, S., 2012. Sowing of low and high diversity seed mixtures in ecological restoration of surface mined-land. Appl. Veg. Sci. 15, 198–207.

Kirwan, L., Lüscher, A., Sebastià, M.T., Finn, J.A., Collins, R.P., Porqueddu, C., Helgadottir, A., Baadshaug, O.H., Brophy, C., Coran, C., Dalmannsdóttir, S., et al., 2007. Evenness drives consistent diversity effects in intensive grassland systems across 28 European sites. J. Ecol. 95, 530–539.

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

Kleijn, D., Bommarco, R., Fijen, T.P., Garibaldi, L.A., Potts, S.G., van der Putten, W.H., 2018. Ecological intensification: bridging the gap between science and practice. Trends Ecol. Evol. 34, 154–166.

Klein, A.M., Steffan-Dewenter, I., Tscharntke, T., 2003. Fruit set of highland coffee increases with the diversity of pollinating bees. Proc. R. Soc. Lond. B Biol. Sci. 270, 955–961.

Knapp, S., van der Heijden, M.G., 2018. A global meta-analysis of yield stability in organic and conservation agriculture. Nat. Commun. 9, 3632.

Koh, L.P., Levang, P., Ghazoul, J., 2009. Designer landscapes for sustainable biofuels. Trends Ecol. Evol. 24, 431–438.

Koh, I., Lonsdorf, E.V., Williams, N., Brittain, C., Isaacs, R., Gibbs, J., Ricketts, T.H., 2016. Modeling the status, trends, and impacts of wild bee abundance in the United States. Proc. Natl. Acad. Sci. 113, 140–145.

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

Kormann, U., Rösch, V., Batáry, P., Tscharntke, T., Orci, K.M., Samu, F., Scherber, C., 2015. Local and landscape management drive trait-mediated biodiversity of nine taxa on small grassland fragments. Divers. Distrib. 21, 1204–1217.

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

Landis, D.A., 2017. Designing agricultural landscapes for biodiversity-based ecosystem services. Basic Appl. Ecol. 18, 1–12.

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

Larsen, T.H., Williams, N.M., Kremen, C., 2005. Extinction order and altered community structure rapidly disrupt ecosystem functioning. Ecol. Lett. 8, 538–547.

Lavorel, S., Grigulis, K., Lamarque, P., Colace, M.P., Garden, D., Girel, J., Pellet, G., Douzet, R., 2011. Using plant functional traits to understand the landscape distribution of multiple ecosystem services. J. Ecol. 99, 135–147.

Lavorel, S., Bayer, A., Bondeau, A., Lautenbach, S., Ruiz-Frau, A., Schulp, N., Seppelt, R., Verburg, P., van Teeffelen, A., Vannier, C., Arneth, A., 2017. Pathways to bridge the biophysical realism gap in ecosystem services mapping approaches. Ecol. Indic. 74, 241–260.

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

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

Lepš, J., 2004. What do the biodiversity experiments tell us about consequences of plant species loss in the real world? Basic Appl. Ecol. 5, 529–534.

Lichtenberg, E.M., Kennedy, C.M., Kremen, C., Batáry, P., Berendse, F., Bommarco, R., Bosque-Pérez, N.A., Carvalheiro, L.G., Snyder, W.E., Williams, N.M., Winfree, R., 2017. A global synthesis of the effects of diversified farming systems on arthropod diversity within fields and across agricultural landscapes. Glob. Chang. Biol. 23, 4946–4957.

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

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

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

Loreau, M., Mouquet, N., Gonzalez, A., 2003. Biodiversity as spatial insurance in heterogeneous landscapes. Proc. Natl. Acad. Sci. 100, 12765–12770.

Lundholm, J., MacIvor, J.S., MacDougall, Z., Ranalli, M., 2010. Plant species and functional group combinations affect green roof ecosystem functions. PLoS One 5, e9677.

Lyons, K.G., Schwartz, M.W., 2001. Rare species loss alters ecosystem function–invasion resistance. Ecol. Lett. 4, 358–365.

Mace, G.M., Norris, K., Fitter, A.H., 2012. Biodiversity and ecosystem services: a multilayered relationship. Trends Ecol. Evol. 27, 19–26.

Maestre, F.T., Quero, J.L., Gotelli, N.J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M.A., Soliveres, S., Escolar, C., 2012. Plant species richness and ecosystem multifunctionality in global drylands. Science 335, 214–218.

Magrini, M.B., Anton, M., Chardigny, J.M., Duc, G., 2018. Pulses for sustainability: breaking agriculture and food sectors out of lock-in. Front. Sust. Food Syst. 2, 64.

Manning, P., Newington, J.E., Robson, H.R., Saunders, M., Eggers, T., Bradford, M.A., Bardgett, R.D., Bonkowski, M., Ellis, R.J., Gange, A.C., 2006. Decoupling the direct and indirect effects of nitrogen deposition on ecosystem function. Ecol. Lett. 9, 1015–1024.

Manning, P., Vries, F.T., Tallowin, J.R., Smith, R., Mortimer, S.R., Pilgrim, E.S., Harrison, K.A., Wright, D.G., Quirk, H., Benson, J., Shipley, B., et al., 2015. Simple measures of climate, soil properties and plant traits predict national-scale grassland soil carbon stocks. J. Appl. Ecol. 52, 1188–1196.

Manning, P., Plas, F., Soliveres, S., Allan, E., Maestre, F.T., Mace, G., Whittingham, M.J., Fischer, M., 2018. Redefining ecosystem multifunctionality. Nat. Ecol. Evol. 2, 427.

Maron, J.L., Marler, M., Klironomos, J.N., Cleveland, C.C., 2011. Soil fungal pathogens and the relationship between plant diversity and productivity. Ecol. Lett. 14, 36–41.

Martín-López, B., Iniesta-Arandia, I., García-Llorente, M., Palomo, I., Casado-Arzuaga, I., Del Amo, D.G., Gómez-Baggethun, E., Oteros-Rozas, E., Palacios-Agundez, I., Willaarts, B., González, J.A., 2012. Uncovering ecosystem service bundles through social preferences. PLoS One 7, e38970.

Maskell, L.C., Crowe, A., Dunbar, M.J., Emmett, B., Henrys, P., Keith, A.M., Norton, L.R., Scholefield, P., Clark, D.B., Simpson, I.C., Smart, S.M., Clough, Y., 2013. Exploring the ecological constraints to multiple ecosystem service delivery and biodiversity. J. Appl. Ecol. 50, 561–571.

Meyer, S.T., Ptacnik, R., Hillebrand, H., Bessler, H., Buchmann, N., Ebeling, A., Eisenhauer, N., Engels, C., Fischer, M., Halle, S., Klein, A.M., et al., 2018. Biodiversity–multifunctionality relationships depend on identity and number of measured functions. Nat. Ecol. Evol. 2, 44.

Mori, A.S., Isbell, I., Seidl, R., 2018. β–diversity, community assembly, and ecosystem functioning. Trends Ecol. Evol. 33, 549–564.

Nyfeler, D., Huguenin-Elie, O., Suter, M., Frossard, E., Connolly, J., Lüscher, A., 2009. Strong mixture effects among four species in fertilized agricultural grassland led to persistent and consistent transgressive overyielding. J. Appl. Ecol. 46, 683–691.

Oehri, J., Schmid, B., Schaepman-Strub, G., Niklaus, P.A., 2017. Biodiversity promotes primary productivity and growing season lengthening at the landscape scale. Proc. Nat. Acad. Sci. 114, 10160–10165.

Oliver, T.H., Heard, M.S., Isaac, N.J., Roy, D.B., Procter, D., Eigenbrod, F., Freckleton, R., Hector, A., Orme, C.D.L., Petchey, O.L., 2015. Biodiversity and resilience of ecosystem functions. Trends Ecol. Evol. 30, 673–684.

Otway, S.J., Hector, A., Lawton, J.H., 2005. Resource dilution effects on specialist insect herbivores in a grassland biodiversity experiment. J. Anim. Ecol. 74, 234–240.

Pan, Q., Tian, D., Naeem, S., Auerswald, K., Elser, J.J., Bai, Y., Huang, J., Wang, Q., Wang, H., Wu, J., Han, X., 2016. Effects of functional diversity loss on ecosystem functions are influenced by compensation. Ecology 97, 2293–2302.

Peeters, A., Beaufoy, G., Canals, R.M., de Vliegher, A., Huyghe, C., Isselstein, J., Jones, G., Kessler, W., Kirilov, A., Mosquera-Losada, M.R., et al., 2014. Grassland term definitions and and classifications adapted to the delivery of European grassland-based systems. Grassl. Sci. Eur. 19, 743–750.

Pretzsch, H., Schütze, G., 2009. Transgressive overyielding in mixed compared with pure stands of Norway spruce and European beech in Central Europe: evidence on stand level and explanation on individual tree level. Eur. J. For. Res. 128, 183–204.

Rader, R., Birkhofer, K., Schmucki, R., Smith, H.G., Stjernman, M., Lindborg, R., 2014. Organic farming and heterogeneous landscapes positively affect different measures of plant diversity. J. Appl. Ecol. 51, 1544–1553.

Raseduzzaman, M., Jensen, E.S., 2017. Does intercropping enhance yield stability in arable crop production? A meta-analysis. Eur. J. Agron. 91, 25–33.

Redlich, S., Martin, E.A., Steffan-Dewenter, I., 2018. Landscape-level crop diversity benefits biological pest control. J. Appl. Ecol. 55, 2419–2428.

Reich, P.B., Knops, J., Tilman, D., Craine, J., Ellsworth, D., Tjoelker, M., Lee, T., Wedink, D., Naeem, S., Bahauddin, D., et al., 2001. Plant diversity enhances ecosystem responses to elevated CO_2 and nitrogen deposition. Nature 410, 809–812.

Reiss, E.R., Drinkwater, L.E., 2018. Cultivar mixtures: a meta-analysis of the effect of intraspecific diversity on crop yield. Ecol. Appl. 28, 62–77.

Rosa-Scleich, J., Loos, J., Musshoff, O., Tscharntke, T., 2019. Ecological-economic trade-offs of diversified farming systems—a review. Ecol. Econ. 160, 251–263.

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

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

Roschewitz, I., Gabriel, D., Tscharntke, T., Thies, C., 2005. The effects of landscape complexity on arable weed species diversity in organic and conventional farming. J. Appl. Ecol. 2005 (42), 873–882.

Royal Society for the Protection of Birds (RSPB), Hope Farm, 2019. https://www.rspb.org.uk/our-work/conservation/conservation-and-sustainability/farming/hope-farm/.

Samnegard, U., Alins, G., Boreux, V., Bosch, J., García, D., Happe, A.-K., Klein, A.M., Miñarro, M., Mody, K., Porcel, M., et al., 2019. Management trade-offs on ecosystem services in apple orchards across Europe: direct and indirect effects of organic production. J. Appl. Ecol. 56, 802–811.

Scherber, C., Eisenhauer, N., Weisser, W.W., Schmid, B., Voigt, W., Fischer, M., Schulze, E.-D., Roscher, C., Weigelt, A., Allan, E., et al., 2010. Bottom-up effects of plant diversity on multitrophic interactions in a biodiversity experiment. Nature 468, 553.

Schmid, B., Hector, A., 2004. The value of biodiversity experiments. Basic Appl. Ecol. 5, 535–542.

Schmid, B., Hector, A., Huston, M.A., Inchausti, P., Nijs, I., Leadley, P.W., Tilman, D., 2002. The design and analysis of biodiversity experiments. In: Biodiversity and Ecosystem Functioning: Synthesis and Perspectives. Oxford University Press, Oxford, pp. 61–75.

Schnitzer, S.A., Klironomos, J.N., HilleRisLambers, J., Kinkel, L.L., Reich, P.B., Xiao, K., Rillig, M.C., Sikes, B.A., Callaway, R.M., Mangan, S.A., 2011. Soil microbes drive the classic plant diversity–productivity pattern. Ecology 92, 296–303.

Schulze, E.-D., Mooney, H.A., 1994. Ecosystem function of biodiversity: a summary. In: Biodiversity and Ecosystem Function. Springer, pp. 497–510.

Smith, M.D., Knapp, A.K., 2003. Dominant species maintain ecosystem function with non-random species loss. Ecol. Lett. 6, 509–517.

Smith, R., Shiel, R., Bardgett, R.D., Millward, D., Corkhill, P., Evans, P., Quirk, H., Hobbs, P., Kometa, S., 2008. Long-term change in vegetation and soil microbial communities during the phased restoration of traditional meadow grassland. J. Appl. Ecol. 45, 670–679.

Soliveres, S., Manning, P., Prati, D., Gossner, M.M., Alt, F., Arndt, H., Baumgartner, V., Binkenstein, J., Birkhofer, K., Blaser, S., et al., 2016a. Locally rare species influence grassland ecosystem multifunctionality. Philos. Trans. R. Soc. B 371, 20150269.

Soliveres, S., Van Der Plas, F., Manning, P., Prati, D., Gossner, M.M., Renner, S.C., Alt, F., Arndt, H., Baumgartner, V., Binkenstein, J., et al., 2016b. Biodiversity at multiple trophic levels is needed for ecosystem multifunctionality. Nature 536, 456.

Srivastava, D.S., Vellend, M., 2005. Biodiversity-ecosystem function research: is it relevant to conservation? Annu. Rev. Ecol. Evol. Syst. 36, 267–294.

Stein, C., Auge, H., Fischer, M., Weisser, W.W., Prati, D., 2008. Dispersal and seed limitation affect diversity and productivity of montane grasslands. Oikos 117, 1469–1478.

Storkey, J., Döring, T., Baddeley, J., Collins, R., Roderick, S., Jones, H., Watson, C., 2015. Engineering a plant community to deliver multiple ecosystem services. Ecol. Appl. 25, 1034–1043.

Suding, K.N., Ashton, I.W., Bechtold, H., Bowman, W.D., Mobley, M.L., Winkleman, R., 2008. Plant and microbe contribution to community resilience in a directionally changing environment. Ecol. Monogr. 78, 313–329.

Suter, M., Connolly, J., Finn, J.A., Loges, R., Kirwan, L., Sebastià, M.T., Lüscher, A., 2015. Nitrogen yield advantage from grass–legume mixtures is robust over a wide range of legume proportions and environmental conditions. Glob. Chang. Biol. 21, 2424–2438.

Suter, D., Rosenberg, E., Mosimann, E., Frick, R., 2017. Standardmischungen für den Futterbau Revision 2017–2020. Agrarforschung Schweiz 8, 1–16.

Tallis, H., 2011. Natural Capital: Theory and Practice of Mapping Ecosystem Services. Oxford University Press.

Teuscher, M., Gérard, A., Brose, U., Buchori, D., Clough, Y., Ehbrecht, M., Hölscher, D., Irawan, B., Sundawati, L., Wollni, M., Kreft, H., 2016. Experimental biodiversity enrichment in oil–palm-dominated landscapes in Indonesia. Front. Plant Sci. 7, 1538.

Tilman, D., Wedin, D., Knops, J., 1996. Productivity and sustainability influenced by biodiversity in grassland ecosystems. Nature 379, 718.

Tilman, D., Reich, P.B., Knops, J., Wedin, D., Mielke, T., Lehman, C., 2001. Diversity and productivity in a long-term grassland experiment. Science 294, 843–845.

Tilman, D., Hill, J., Lehman, C., 2006. Carbon–negative biofuels from low-input high-diversity grassland biomass. Science 314, 1598–1600.

Tscharntke, T., Klein, A.M., Kruess, A., Steffan-Dewenter, I., Thies, C., 2005. Landscape perspectives on agricultural intensification and biodiversity–ecosystem service management. Ecol. Lett. 8, 857–874.

Tscharntke, T., Tylianakis, J.M., Rand, T.A., Didham, R.K., Fahrig, L., Batáry, P., Bengtsson, J., Clough, Y., Crist, T.O., Dormann, C.F., Ewers, R.M., 2012. Landscape moderation of biodiversity patterns and processes-eight hypotheses. Biol. Rev. 87, 661–685.

Tscharntke, T., Karp, D.S., Chaplin-Kramer, R., Batáry, P., DeClerck, F., Gratton, C., Hunt, L., Ives, A., Jonsson, M., Larsen, A., Martin, E.A., 2016. When natural habitat fails to enhance biological pest control–five hypotheses. Biol. Conserv. 204, 449–458.

UK National Ecosystem Assessment, 2011. The UK National Ecosystem Assessment: Synthesis of the Key Findings. UNEP-WCMC, Cambridge.

van der Plas, F., 2019. Biodiversity and ecosystem functioning in naturally assembled communities. Biol. Rev. https://doi.org/10.1111/brv.12499.

van der Plas, F., Manning, P., Soliveres, S., Allan, E., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M.A., Ampoorter, E., Baeten, L., Barbaro, L., et al., 2016. Biotic homogenization can decrease landscape-scale forest multifunctionality. Proc. Natl. Acad. Sci. 113, 3557–3562.

van der Plas, F., Ratcliffe, S., Ruiz-Benito, P., Scherer-Lorenzen, M., Verheyen, K., Wirth, C., Zavala, M.A., Ampoorter, E., Baeten, L., Barbaro, L., Bastias, C.C., et al., 2018. Continental mapping of forest ecosystem functions reveals a high but unrealised potential for forest multifunctionality. Ecol. Lett. 21, 31–42.

van der Plas, F., Allan, E., Fischer, M., Alt, F., Arndt, H., Binkenstein, J., Blaser, S., Blüthgen, N., Böhm, S., Hölzel, N., Klaus, V.H., et al., 2019. Towards the development of general rules describing landscape heterogeneity–multifunctionality relationships. J. Appl. Ecol. 56, 168–179.

Van der Putten, W.H., Mortimer, S.R., Hedlund, K., Van Dijk, C., Brown, V.K., Lepä, J., Rodriguez-Barrueco, C., Roy, J., Len, T.D., Gormsen, D., Korthals, G.W., 2000. Plant species diversity as a driver of early succession in abandoned fields: a multi-site approach. Oecologia 124, 91–99.

Vandermeer, J.H., 1992. The Ecology of Intercropping. Cambridge University Press.

Vellend, M., Baeten, L., Myers-Smith, I.H., Elmendorf, S.C., Beauséjour, R., Brown, C.D., De Frenne, P., Verheyen, K., Wipf, S., 2013. Global meta-analysis reveals no net change in local-scale plant biodiversity over time. Proc. Natl. Acad. Sci. 110, 19456–19459.

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

Vogel, A., Ebeling, A., Gleixner, G., Roscher, C., Scheu, S., Ciobanu, M., Koller-France, E., Lange, M., Lochner, A., Meyer, S.T., et al., 2019. A new experimental approach to test why biodiversity effects strengthen as ecosystems age. Adv. Ecol. Res. this issue, Chapter 7.

Walter, A., Finger, R., Huber, R., Buchmann, N., 2017. Opinion: smart farming is key to developing sustainable agriculture. Proc. Natl. Acad. Sci. 114, 6148–6150.

Wardle, D.A., 2016. Do experiments exploring plant diversity–ecosystem functioning relationships inform how biodiversity loss impacts natural ecosystems? J. Veg. Sci. 27, 646–653.

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

Weisser, W.W., Roscher, C., Meyer, S.T., Ebeling, A., Luo, G., Allan, E., Beßler, H., Barnard, R., Buchmann, N., Buscot, F., Engels, C., et al., 2017. Biodiversity effects on ecosystem functioning in a 15-year grassland experiment: patterns, mechanisms, and open questions. Basic Appl. Ecol. 23, 1–73.

Werling, B.P., Dickson, T.L., Isaacs, R., Gaines, H., Gratton, C., Gross, K.L., Liere, H., Malmstrom, C.M., Meehan, T.D., Ruan, L., Robertson, B.A., et al., 2014. Perennial grasslands enhance biodiversity and multiple ecosystem services in bioenergy landscapes. Proc. Natl. Acad. Sci. 111, 41652–41657.

Wilson, J.B., Peet, R.K., Dengler, J., Pärtel, M., 2012. Plant species richness: the world records. J. Veg. Sci. 23, 796–802.

Winfree, R., Jeremy, W.F., Williams, N.M., Reilly, J.R., Cariveau, D.P., 2015. Abundance of common species, not species richness, drives delivery of a real-world ecosystem service. Ecol. Lett. 18, 626–635.

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

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

Wright, A.J., Wardle, D.A., Callaway, R., Gaxiola, A., 2017. The overlooked role of facilitation in biodiversity experiments. Trends Ecol. Evol. 32, 383–390.

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

Yang, Y., Tilman, D., Furey, G., Lehman, C., 2019. Soil carbon sequestration accelerated by restoration of grassland biodiversity. Nat. Commun. 10, 718.

Zavaleta, E.S., Hulvey, K.B., 2004. Realistic species losses disproportionately reduce grassland resistance to biological invaders. Science 306, 1175–1177.

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

Zhu, J., Jiang, L., Zhang, Y., 2016. Relationships between functional diversity and above-ground biomass production in the northern Tibetan alpine grasslands. Sci. Rep. 6, 34105.

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

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