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Quantifying farm sustainability through the lens of ecological theory

Jonathan Storkey[1,*], Chloe Maclaren[1,2], James M. Bullock[3], Lisa R. Norton[4], John W. Redhead[3] and Richard F. Pywell[3]

[1]*Protecting Crops and the Environment, Rothamsted Research, West Common, Harpenden, Hertfordshire, AL5 2JQ, UK*
[2]*Department of Crop Production Ecology, Swedish University of Agricultural Sciences, Almas Alle 8, Uppsala 750 07, Sweden*
[3]*UK Centre for Ecology & Hydrology (UKCEH), Maclean Building, Benson Lane, Wallingford, OX10 8BB, UK*
[4]*UKCEH, Lancaster Environment Centre, Lancaster, LA1 4AP, UK*

ABSTRACT

The achievements of the Green Revolution in meeting the nutritional needs of a growing global population have been won at the expense of unintended consequences for the environment. Some of these negative impacts are now threatening the sustainability of food production through the loss of pollinators and natural enemies of crop pests, the evolution of pesticide resistance, declining soil health and vulnerability to climate change. In the search for farming systems that are sustainable both agronomically and environmentally, alternative approaches have been proposed variously called 'agroecological', 'conservation agriculture', 'regenerative' and 'sustainable intensification'. While the widespread recognition of the need for more sustainable farming is to be welcomed, this has created etymological confusion that has the potential to become a barrier to transformation. There is a need, therefore, for objective criteria to evaluate alternative farming systems and to quantify farm sustainability against multiple outcomes. To help meet this challenge, we reviewed the ecological theories that explain variance in regulating and supporting ecosystem services delivered by biological communities in farmland to identify guiding principles for management change. For each theory, we identified associated system metrics that could be used as proxies for agroecosystem function. We identified five principles derived from ecological theory: (*i*) provide key habitats for ecosystem service providers; (*ii*) increase crop and non-crop habitat diversity; (*iii*) increase edge density: (*iv*) increase nutrient-use efficiency; and (*v*) avoid extremes of disturbance. By making published knowledge the foundation of the choice of associated metrics, our aim was to establish a broad consensus for their use in sustainability assessment frameworks. Further analysis of their association with farm-scale data on biological communities and/or ecosystem service delivery would provide additional validation for their selection and support for the underpinning theories.

Key words: regenerative farming, agroecology, ecosystem services, natural capital, integrated pest management.

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* Author for correspondence (Tel.: (+44) 1582 938550; E-mail: jonathan.storkey@rothamsted.ac.uk).

I. INTRODUCTION

(1) Current threats to the sustainability of farming

Agricultural landscapes across the developed world are dominated by farming systems with a common lineage that reflects the demands of a growing global population and the strong policy drive, especially in Europe, for national food security and affordable food following the privations of two world wars. Post-war advances in crop breeding led by the Nobel prize-winning laureate, Norman Borlaug (Swaminathan, 2009), the invention of agrochemicals (inorganic fertilisers and pesticides) and increased mechanisation of farming operations, hereafter, the 'Green Revolution' (Evenson & Gollin, 2003), have led to the convergence of farming systems around four common properties: (*i*) specialisation of business enterprises (away from mixed systems to exclusively livestock or arable); (*ii*) investment in a small number of crop and livestock species and genotypes supported by large agribusiness and global markets; (*iii*) increased scale of production both in terms of the size of individual land parcels and the overall enterprise; and (*iv*) reliance on external inputs of fertilisers and chemical crop protection products (hereafter, 'pesticides' – including herbicides, insecticides and fungicides) and of hydrocarbons to fuel mechanisation. In many areas of the developed world, the resulting homogenisation of farming systems and landscapes has largely replaced the more diverse, locally adapted, small-scale farms that characterised those landscapes up to the middle of the last Century.

The success of the Green Revolution has undoubtedly brought dramatic benefits for humankind and has delivered on the policy goal of ensuring food production keeps pace with a growing global population (Evenson & Gollin, 2003). However, this success has been accompanied by well documented negative unintended consequences for the environment and society. These can be summarised as: (*a*) the large-scale simplification of landscapes negatively impacting national and regional-scale biodiversity through the loss of semi-natural habitats resulting in biotic homogenisation and the selection of generalist (including crop pest) species at the expense of specialists (Carvalheiro *et al.*, 2013; Smart *et al.*, 2006). (*b*) Intensive use of agrochemicals and homogenisation of cropping systems leading to declines in the biodiversity of the flora and fauna adapted to ruderal habitats through direct effects of pesticides or loss of resources provided by non-crop plants (Donald *et al.*, 2006; Goulson *et al.*, 2015; Storkey *et al.*, 2012). (*c*) More frequent, mechanised tillage, intensive use of inorganic fertilisers and reduced organic inputs degrading agricultural soils through erosion, contamination, acidification, salinisation, and loss of biological diversity (Kopittke *et al.*, 2019) with particular concern around the impact of declines in soil organic carbon on soil function (Prout *et al.*, 2020). (*d*) Reliance on pesticides for crop protection and simplification of crop rotations selecting for a small species pool of pernicious pests, weeds and diseases and the overuse of a limited number of pesticide active ingredients leading to the evolution of pesticide resistance (Hawkins *et al.*, 2019). (*e*) Losses of pesticides and chemical fertilisers through volatilisation, leaching or adsorption on transported soil particles polluting air and water courses with implications for the health of ecosystems and humans (Bell *et al.*, 2021; de Souza *et al.*, 2020). (*f*) The contribution of agriculture to greenhouse gas emissions through land-use change, the reliance on fossil fuels in mechanical operations and the production of inorganic fertilisers plus the emission of nitrous oxides and methane from fertilisers and livestock (Lamb *et al.*, 2021).

The impacts of some of these negative unintended consequences of the Green Revolution (*a*, *e* and *f*) are largely external to the production system and manifested beyond the physical boundaries of the farm. However, others (*b*, *c* and *d*) represent intrinsic threats to the continued production of high-yielding crops by compromising the inherent productivity of the farmed environment. The loss of farmland functional biodiversity (*b*) has compromised 'regulating' ecosystem services (ESs) including pest control and pollination on which crop production relies (Bommarco, Kleijn & Potts, 2013). Declining soil organic carbon (*c*) has negatively impacted supporting services of soil water-holding capacity and nutrient cycling by soil microbes (Neal *et al.*, 2020).

Finally, the evolution of pesticide resistance (*d*) has had a significant impact on farm productivity and profits (Varah *et al.*, 2020). These stresses on the system threaten the agronomic sustainability of farming and may partly explain the widening 'yield gap' between the genetic potential of modern crop cultivars and realised on-farm yields (van Ittersum *et al.*, 2013). The remaining unintended consequences of the Green Revolution that are external to the production system can also be perceived as indirect threats to agronomic sustainability. The simplification of agricultural landscapes (*a*) reduces species pools of beneficial organisms that might impact long-term resilience of agricultural systems to environmental shocks (Oliver *et al.*, 2015; Redhead *et al.*, 2018). Losses of fertilisers and pesticides to the environment (*e*) creates pressure on policymakers to tighten the regulatory framework for agrochemical use, reducing their use as agronomic management tools (for example, through the Green Deal and the Farm to Fork Strategy, the European Union currently has the ambitious goal to reduce pesticide use by 50% by 2030). Finally, climate change, driven in part by agricultural emissions (*f*) is bringing about a wide variety of risks and challenges to production systems (Ortiz-Bobea *et al.*, 2021) and the drive for net zero is creating additional pressure to reduce inputs of inorganic fertilisers.

In response to these global threats, there is now a growing political and scientific consensus that the current, dominant paradigm of 'industrialised' food production is not sustainable (Pingali, 2012; Godfray *et al.*, 2010) – where 'sustainability' is here defined as 'meeting the needs of the present without compromising the ability of future generations to meet their own needs' (Brundtland, 1987, p. 16). If we assume that the primary purpose of farms will remain (in the medium term) the provision of food, then a sustainable farm can be defined as one that can continue to deliver this 'provisioning' ES, *sensu* the Millennium Ecosystem Assessment (2005), at a level that meets the nutritional needs of future generations without compromising the associated regulating, supporting and cultural ESs delivered to society by the farm or the surrounding landscape (Garnett *et al.*, 2013).

(2) The problem of measuring farm sustainability

In the face of the incontrovertible evidence of the negative environmental impact of large-scale, industrialised farming and the associated threats to sustainable production, there is movement (particularly in Europe) towards a transformation of farming systems. This is more than the search for a new technological breakthrough (such as gene editing) that could catalyse a 'second Green Revolution' but rather an acknowledgment that a more holistic approach is required that maintains both production and the functional integrity of the agroecosystem by investing in natural capital and ESs (Jones *et al.*, 2016). Over time, terms used to describe this alternative approach to farming have proliferated – we identified a list of eight commonly used terms in the sustainable agriculture literature (Fig. 1). At the time of writing, the term 'regenerative farming/agriculture' is increasingly used. While the widespread recognition of the need for more sustainable farming is to be welcomed, this has created etymological confusion that has the potential to become a barrier to transformation, particularly at the level of the practitioner, and may also limit cooperation and knowledge sharing between groups espousing different terms. It also raises the problem of 'categorisation'. Every individual farm is a unique combination of landscape and environmental context and management options implemented at multiple spatial and temporal scales. While it is possible to generalise farm types in terms of broad environmental constraints and farming systems (Goodwin *et al.*, 2022; Rodríguez, van

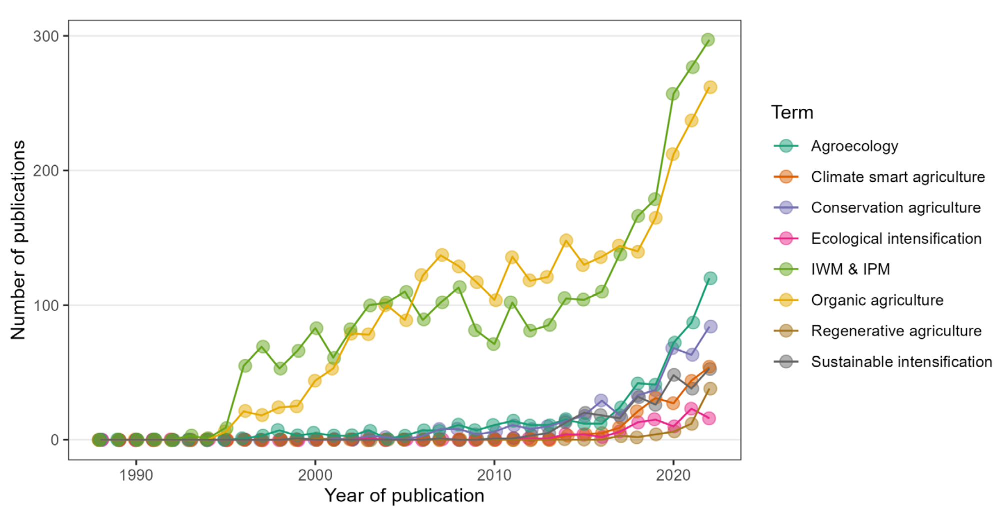

Fig. 1. Number of publications identified by a *Web of Science* search using eight search terms associated with the concept of sustainable farming, illustrating the diversity of terminologies. 'Organic' and 'integrated pest management (IPM)' are well-established terms, other concepts, including 'agroecology' and 'regenerative agriculture' are becoming increasingly used but are less well defined. IWM, integrated weed management.

Bussel & Alkemade, 2024), objectively categorising farms according to the more subtle management criteria intrinsic to terms included in Fig. 1 is much more challenging. With the exception of certification schemes with prescribed criteria (for example to qualify produce as 'organic'), there are no agreed criteria for defining farms as, for example, 'regenerative or 'agroecological'.

What is needed, therefore, is a robust, evidence-based protocol for benchmarking the sustainability of farms in terms of multiple outcomes and monitoring the impact of change on continuous gradients of system properties related to environmental and agronomic sustainability. This represents a significant challenge compared to assessing the past success of the Green Revolution that had a single, clearly articulated outcome (increased yield of major crops) that could be easily measured, meaning the relative benefit of any change in practice could be assessed based on its impact on that single outcome (Evenson & Gollin, 2003). There are now several examples in the literature of proposed lists of metrics or indicators that can be used to quantify multiple outcomes and could form the basis of an assessment framework for farm sustainability (Bockstaller *et al.*, 2008; Bonisoli, Galdeano-Gomez & Piedra-Munoz, 2018; Gharsallah, Gandolfi & Facchi, 2021; Mahon *et al.*, 2018; Smith *et al.*, 2017). For such an assessment framework to be relevant at the level of an individual farm, any proposed metrics or indicators must: (*i*) be measurable in a cost effective and practical way; (*ii*) relate to clearly defined outcomes; and (*iii*) be sensitive to management change so that the impact of an intervention on outcomes can be predicted. However, a recent review of alternative lists of indicators concluded that the lack of consensus amongst experts and transparency in the logic behind the criteria for their selection is currently an impediment to the implementation of a robust framework of this type (de Olde *et al.*, 2017).

(3) Guiding principles for linking management to ecosystem services

Against this background, instead of starting with the question of 'what to measure?', we first review the established knowledge on the ecological processes that underpin the sustainability of both food and wider ES production on farmland. Candidate system properties that can serve as proxies for the status of this ecological function are then identified with reference to guiding principles for beneficial management change. For the purposes of this review, our focus is on agronomic and environmental sustainability with an emphasis on the regulating and supporting ESs delivered by biological communities on farms: (*i*) regulation of pest and weed populations including the predation of invertebrate pests and weed seeds by natural enemy communities; (*ii*) pollination of insect-pollinated crops; and (*iii*) nutrient cycling and water regulation in soils by microbial communities and meso/macrofauna.

In turn, these will determine the downstream outcomes of sustainable food production and agrochemical use and mitigate the threats to production associated with the unintended consequences of the Green Revolution that threaten agronomic sustainability. We acknowledge the landscape and community ecology bias in our review of ecological theory which we do not intend to be comprehensive. In every case, however, the associated metrics we identify are also relevant to assessing impacts of agriculture on wider environmental sustainability beyond the boundaries of the farm and production system and these are also discussed. The fundamental understanding of the relationship between ESs and farm-scale metrics also facilitates prediction of the impact of a management change if the metrics are also sensitive to specific options (Fig. 2). These management options can be thought of as the 'ingredients' that are common to the alternative 'recipes' for different sustainable farming approaches included in Fig. 1. Rather than attempting to quantify the direct effects of individual management changes (or combinations thereof) on ES, our approach is therefore to predict how a management change would be expected to 'nudge' a farm-scale metric as a proxy for ES delivery.

Based on a knowledge of the literature on the ecological processes that determine the functioning of agroecosystems and underpin crop production, we identified five ecological theories that are useful for identifying sustainability metrics. In each case, starting with an introduction to the ecological theory, we use the theory to suggest a guiding principle to inform practice before identifying potential properties of a farm that could be used as metrics. These could be used in a framework for benchmarking and monitoring farms in terms of the delivery and resilience of ESs related to agronomic sustainability. Finally, in each case, the management interventions that we predict would have a positive benefit on the ES, reflected by a change in the metrics, are discussed.

II. FIVE ECOLOGICAL THEORIES UNDERPINNING FARM SUSTAINABILITY

(1) Mass ratio hypothesis: maintain sufficient areas of habitat on farms to provide resources for functionally important ES providers

The 'mass ratio hypothesis' proposed by Grime (1998) states that ecosystem function is primarily determined by the traits of the dominant species in a biological community with evidence for the community weighted mean of 'effect traits' (*sensu* Lavorel, 2013) being more important than trait divergence in determining ES delivery (Diaz *et al.*, 2007). In practice, this means that the delivery of a given ES on farmland will be weighted by the abundance of individuals with traits that result in them making a disproportionate contribution to ES delivery. Hence the status of their populations (expressed either at a species or functional-group level) or response to management change is indicative of potential effects on ESs. This theory is most relevant to maintaining the regulating ESs of pollination (Potts *et al.*, 2010) and predation of weed seeds and crop pests (Begg *et al.*, 2017);

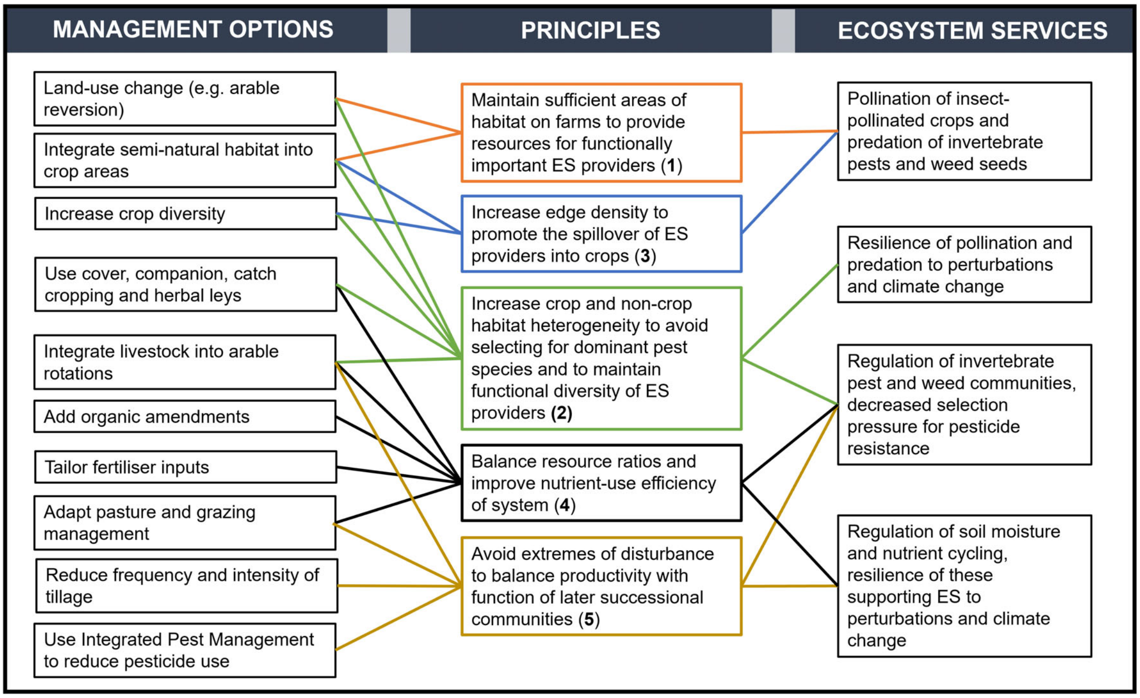

Fig. 2. The impact of potential management options on ecosystem services (ESs) interpreted in terms of their contribution to meeting five guiding principles based on ecological theories. The ecological theories on which the principles are based (numbers in parentheses) are discussed in detail in Section II; in each case a farm-scale metric is also proposed to monitor and benchmark progress based on continuous gradients (summarised in Section III).

the latter providing 'top-down' regulation of pest and weed populations and reducing the frequency with which pesticides need to be applied to crops (by using surveillance and use of thresholds; Barzman *et al.*, 2015). This will, in turn, reduce non-target impacts as well as the likelihood of resistance evolving – improving the agronomic sustainability of the system.

A literature review of the relationship between invertebrate functional traits and ES delivery, albeit not exclusively in agricultural landscapes, identified specialisation, body size, feeding habit and dispersal range as important traits for both pollination and predation with large, generalist species with wide dispersal ranges (including the examples in Fig. 3) being more effective (de Bello *et al.*, 2010). There is evidence that generalist species are also more resilient to environmental perturbations (Redhead *et al.*, 2018). While it is possible to identify and measure abundance of these species or their functional groups at the farm scale directly (by using, for example pollinator transects or pitfall trapping), this relies on experts and is not easily scalable. An alternative is to assess habitat provision on a farm in terms of the ecological requirements of ES providers (Butler *et al.*, 2009; Staley *et al.*, 2021), or the overlap of invertebrate effect traits with the response traits of the plant communities that support them (Lavorel *et al.*, 2013). For example, for pollinators, farmland habitats could be evaluated in terms of the spatiotemporal provision of pollen and nectar using vegetation classification schemes and databases of canopy architecture, flowering time and flower traits (Baude *et al.*, 2016) and, for predators by using habitat characteristics that indicate essential refugia outside the crop or cropping season (Woodcock *et al.*, 2010).

There is extensive evidence in the literature for the importance of the absolute amount of semi-natural habitat in the landscape for supporting ESs on farmland; often quantified as % semi-natural habitat *versus* % farmed (Tscharntke *et al.*, 2005; Garibaldi *et al.*, 2021; Holland *et al.*, 2015) and a recent study used variants of this metric to explain variance in the resilience of crop yields (Redhead *et al.*, 2020). Weighting semi-natural habitats by their functional contribution to regulating ESs (or quantifying important habitats separately) would provide a useful, complementary way of evaluating alternative approaches to managing the farm landscape to ensure sufficient provision of important resources for ES providers at the farm scale. The literature on the ecological requirements of regulating ES providers suggests that three non-crop farmland habitats are particularly important: areas that provide floral resources (especially in spring) (Carvell *et al.*, 2017; Albrecht *et al.*, 2016), tussocky grass margins (Woodcock *et al.*, 2005) and hedgerows (Montgomery, Caruso & Reid, 2020; Albrecht *et al.*, 2016). Where there are opportunities to increase in-field diversity (see Section II.2), these resources may also be provided by areas of the farm used for production, for example by integrating flowering forbs into grass swards (Woodcock *et al.*, 2014). The amounts of functionally important habitats could be integrated at the farm scale by taking the limiting-factor

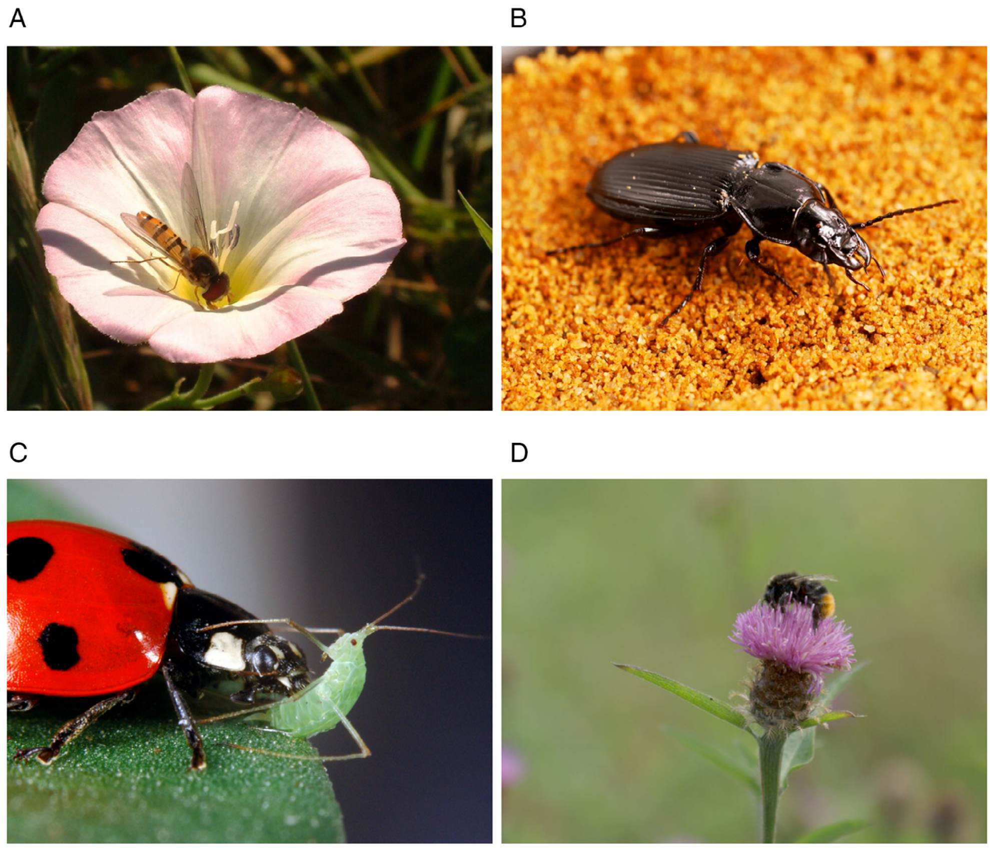

Fig. 3. Examples of generalist species with functional traits that mean they make a disproportionate contribution to pest and weed control in Europe: the marmalade hoverfly (*Episyrphus balteatus*) (A), the predatory ground beetle *Pterostichus melanarius* (B), the seven-spot ladybird (*Coccinella septempunctata*) (C) and for pollination, the red-tailed bumblebee (*Bombus lapidarius*) (D).

approach that uses the sum of reciprocals function (originally used to describe the effects of nutrients on plant growth; Balmukand, 1928). This combines multiple limitations such that the result cannot exceed the smallest component while reflecting the effect of all constraints. Habitat provision at the farm scale could be quantified from high-resolution satellite imagery in combination with or calibrated against habitat surveys (Butcher *et al.*, 2020) and further refined by assigning a measure of habitat quality.

Increasing the absolute amount of non-crop habitat to support regulating and supporting ESs, for example by establishing field margins (Marshall & Moonen, 2002), is the only guiding principle in our list for which there is a potential direct trade-off with food production as it may entail taking land out of cultivation. Although there is recent evidence that the contribution of ESs to increasing yield can compensate for this land taken out of production, especially if established on less-productive parts of the farm (Pywell *et al.*, 2015), there will be thresholds of habitat provision for any given ES below which its delivery is compromised and above which further loss of crop land cannot be justified on the basis of ES provision (Fig. 4). The lower and upper thresholds for these habitats (in terms of the proportion of farm area) will be context specific [depending on variables including farm size and fragmentation of habitat (Carvell *et al.*, 2017)]. While more will generally be better, therefore, we recommend presenting the proportions of functionally important habitats separately when benchmarking farms and relying on expert opinion and local knowledge to identify deficiencies.

(2) Niche partitioning: increase crop and non-crop habitat heterogeneity to avoid selecting for dominant pest species and to maintain functional diversity of ES providers

The relationship between species richness and the delivery of a single ES is case specific and not always positive (Balvanera *et al.*, 2006; Hooper *et al.*, 2005), and mass ratio effects mean that individual services may benefit from an approach that optimises the best-performing functional types rather than species richness (see Section II.1). However, when multiple ESs are assessed in parallel, increasing species richness may be seen as desirable insofar as different species perform complementary functions (Hector & Bagchi, 2007; Zavaleta *et al.*, 2010). Where overlap between species in terms of their contribution to different services is small, the multifunctionality of the system has been predicted to continue to increase as additional species are added to the community. Increasing the diversity of habitats in a landscape will, therefore, provide more ecological niches and support a greater range of

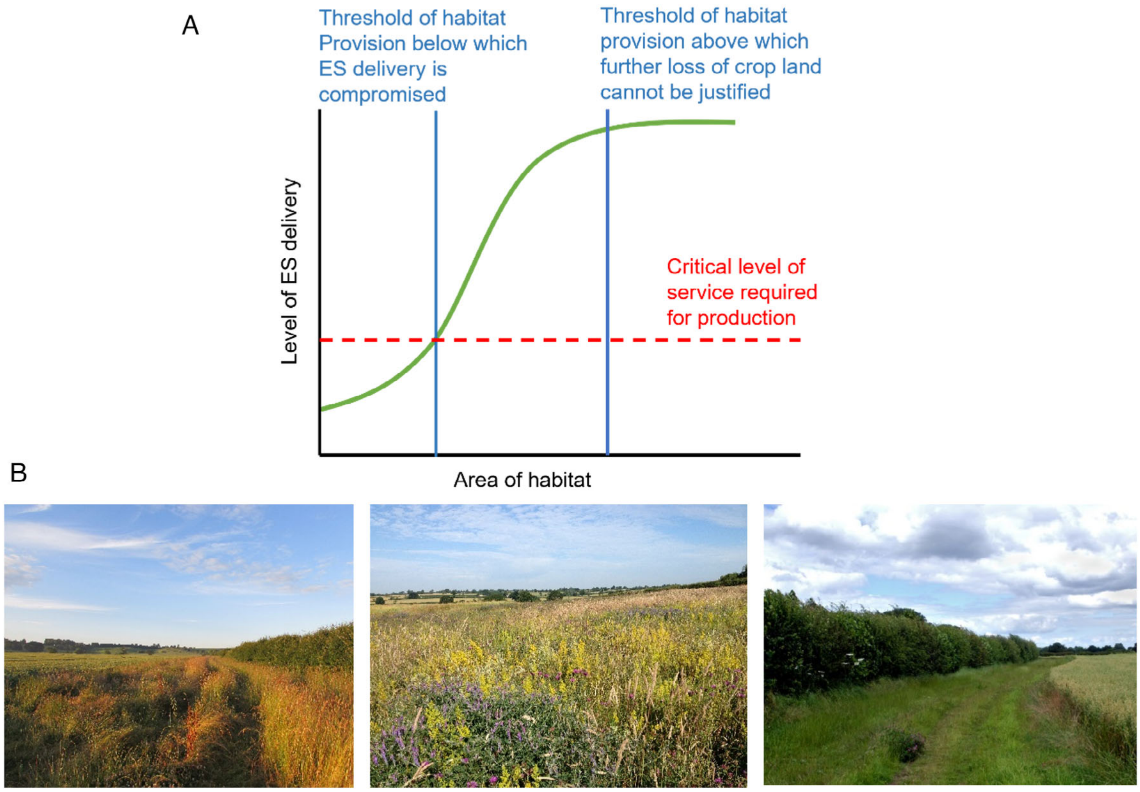

Fig. 4. (A) The proportion of functionally important habitat required to support populations of ecosystem service (ES) providers will be a function of the demand for the ES and the ecological requirements of the target species or functional groups. There will be an upper threshold beyond which further habitat creation cannot be justified based on the need for ES provision. The optimal solution at the farm scale will depend on the ecology of the local species pool, existing provision of resources in the wider landscape, and the precise shape of the curve relating habitat area to service provision. (B) Important non-crop habitats for ES of pollination or predation of invertebrate pests and weed seeds, from left to right: tussocky grass margins, wildflower strips and hedgerows.

biological 'response traits' (*sensu* Lavorel & Garnier, 2002) reflecting different ecological strategies and resource requirements, cascading down to more resilient ES provision (Oliver *et al.*, 2015). In addition, focused studies on individual ESs have also provided some evidence for the potential benefits of biological functional diversity. For example, growing crop species together in an intercrop has multiple benefits for productivity (Tilman, 2020); the resilience of pest control (Greenop *et al.*, 2018) and pollination (Woodcock *et al.*, 2019) is improved by more functionally diverse invertebrate communities; and biodiversity in pastures can contribute to the regulation of soil water (Leimer *et al.*, 2021) and sward nutritional quality (Darch *et al.*, 2020). Such effects have been demonstrated at multiple, potentially interacting, spatial scales, suggesting that habitat heterogeneity provides a useful metric at landscape, farm and field scales (Benton, Vickery & Wilson, 2003).

As well as promoting ecosystem multifunctionality, increasing habitat diversity and niche partitioning in farming systems also disrupts the life cycles of pests, weeds and diseases; a concept that has been understood for millennia and underpins the concept of crop rotations. By favouring or suppressing species with different survival, tolerance, and recovery traits, the greater variety of habitat niches associated with longer rotations reduces the likelihood of pest species becoming dominant and restores diversity to the system (Storkey & Neve, 2018). However, these ecological regulating processes have largely been replaced by pesticides in modern, homogenous cropping systems, with associated threats to sustainability of evolved resistance, loss of biodiversity and negative environmental impacts. The resulting simplification of crop rotations has narrowed the habitat niche of non-crop biodiversity in cultivated fields, selecting for well-adapted, dominant species that have become pernicious pests or weeds of crops; increasing the reliance on specific chemical active ingredients and the likelihood of these species evolving pesticide resistance. Examples include the insect pest cabbage stem flea beetle (*Psylliodes chrysocephala*) in oilseed rape (Willis *et al.*, 2020) and the weed black-grass (*Alopecurus myosuroides*) in wheat (Hicks *et al.*, 2018) (Fig. 5). It can be argued, therefore, when monitored over time, more even plant and insect communities within crop fields are indicative of more sustainable farming systems (Storkey & Neve, 2018).

The simplification of landscapes and crop rotations (including the decline of mixed farming and loss of grass leys from arable systems) has been identified as among the

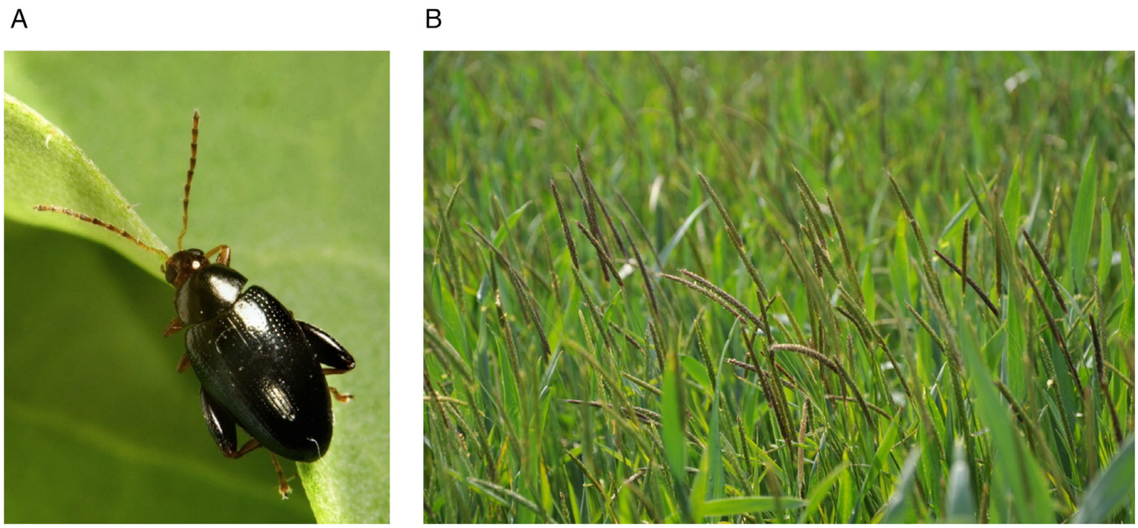

Fig. 5. Examples of a pest and weed species that have been selected for by simplified cropping rotations and have now evolved resistance to pesticides making them serious threats to the sustainable productivity of major crops: cabbage stem flea beetle (*Psylliodes chrysocephala*), a pest of oilseed rape (A) and black-grass (*Alopecurus myosuroides*), a weed of cereal crops (B).

primary drivers of declines in farmland biodiversity, the ESs it delivers and resilience of the agroecosystem system (Benton *et al.*, 2003; Bommarco *et al.*, 2013; Bullock *et al.*, 2017). Measures of taxonomic or functional diversity, dominance or evenness of biological communities on and off fields are therefore useful for evaluating future farm sustainability and the impact of a change in practice. However, as with the abundance of key functional types (see Section II.1) measuring this directly is difficult and habitat diversity is, therefore, a useful proxy. A recent meta-analysis of the effect of different types of agricultural diversification provided convincing evidence for its benefit to the ESs of pollination, pest control, nutrient cycling, soil fertility and water regulation without compromising crop yields (Tamburini *et al.*, 2020). In their analysis, the authors included addition of non-crop habitats within or around the field or in the surrounding landscape as a category of diversification. However, we propose that the ecological principles underlying the impact of crop and non-crop habitats are different, and there is more to be gained by quantifying these separately at the farm scale. The proportion and diversity of non-crop habitats contribute to sustaining abundant and diverse communities of ES providers, while a key additional function of crop diversity is to limit populations of disservice providers and thus regulate pest, weed and disease pressure. That said, it is ultimately the combination of non-crop and crop habitats (and the interfaces between them, see Section II.3) that is important for sustainability at the farm scale and which should be the basis of a comprehensive assessment (see Section III).

Indices of biological diversity, such as the Shannon diversity index or Pielou evenness index, can be applied to the diversity of crops (including pastures) and non-crop habitats separately but require different systems for categorising habitat types. In the case of crop diversity, measured either spatially at the farm scale, temporally at the field rotation level or integrating the two, crop species or functional group (cereal, brassica, legume, root crop, perennial, grass/mixed species leys) can be used (MacLaren *et al.*, 2022). Approaches such as intercropping (growing two or more crops simultaneously on the same field) or undersowing a crop with a living mulch could be categorised as distinct crop types. Existing schemes for classifying non-crop habitats designed for ecological surveys (Butcher *et al.*, 2020) are appropriate for deriving an index of non-crop diversity but would benefit from a finer grained definition based on an understanding of the contribution of different habitats to ecosystem services (Holland *et al.*, 2014). Given the loss of semi-natural habitat on farmland, the simplification of crop rotations and decline in mixed farming since the 1960s, increasing habitat diversity and niche partitioning would generally be expected always to be beneficial in modern farming systems. However, at some point there will be a trade-off with the provision of important non-crop habitats, mean patch size and the area of economically viable cropping that implies there may be an optimum level of non-crop and crop diversity (Maskell *et al.*, 2023).

(3) Spatial mass effects: increase edge density to promote the spillover of ES providers into crops

One explanation for the coexistence of species is the 'spillover' of individuals from habitats with positive growth rates into neighbouring habitats in which their fitness is lower but populations are maintained by frequent recolonisation (Shmida & Wilson, 1985). Alternatively termed 'spatial mass effects' or 'source–sink dynamics', this process contributes to maintaining α (within-habitat diversity) through β (between-habitat) diversity. Through the use of inorganic fertilisers and chemical pesticides and the simplification of crop rotations, α diversity of biological communities adapted to the regularly disturbed environments of cultivated fields has declined dramatically since the advent of the Green Revolution in the 1960s (Butler, Vickery & Norris, 2007). While this is a conservation concern, it also has implications for agronomic sustainability as some of these species also provide the ESs of crop pollination and regulation of crop pests. Modern crop fields are largely deficient in the primary resources provided to these beneficial invertebrates by weeds (which have been removed by herbicides) and there is a growing evidence base for the importance of the spillover of beneficial invertebrates from neighbouring semi-natural habitats to deliver these services (Rand, Tylianakis & Tscharntke, 2006).

A recent meta-analysis of the effect of landscape configuration on functional biodiversity and agroecosystem services established the benefit of increasing 'edge density' in agricultural landscapes in the context of landscapes with varying amounts of seminatural habitat (Martin *et al.*, 2019). Edge density (km/ha) is defined as the total length of edges (or interfaces between land parcels) divided by total area of the landscape, and thus tends to increase as habitat patches become both more dispersed and connected throughout a landscape. Although significant interactions with landscape composition were also observed, edge density consistently explained additional variance in the abundance of pollinators and natural enemy species measured at the landscape scale. One explanation for this result is that a landscape with higher edge density may benefit populations of dispersal-limited species through smaller scale provision of spatially and temporally heterogenous resources. However, spillover effects would also be expected to be more important in maintaining α diversity in these landscapes; potentially increasing yield in fields neighbouring habitats with high abundance of beneficial invertebrates (Pywell *et al.*, 2015). Martin *et al.* (2019) only observed a few examples of a hump-backed relationship between ESs and edge density indicating that edges were generally not acting as barriers to dispersal at the spatial scales analysed. We, therefore, propose that, for benchmarking farms or monitoring change, increasing edge density should generally be viewed positively. However, taxa adapted to open landscapes may respond negatively to the addition of some boundary features (Carrasco *et al.*, 2018; Jonason *et al.*, 2013). We also note that for a given proportion of habitat in a landscape, edge density trades off against habitat patch size (Martin *et al.*, 2019), which may be important to conserve species that are sensitive to edge effects and adapted to large areas of undisturbed habitat (MacLaren, Buckley & Hale, 2014; Phalan *et al.*, 2011).

Edge density can be measured at the farm scale using remote-sensing imagery and increased by introducing additional landscape features such as boundary features (hedges or tree lines), additional patches of semi-natural habitat or in-field strips of non-crop vegetation such as beetle banks or strips of wildflowers. In the analysis of Martin *et al.* (2019), all edges are treated as being equal in terms of explaining variance in ES provision. However, a simple measure of edge density at the farm scale could be weighted by the functional difference between community types, so that, for example, an edge between two cereal crops would make only a minor contribution to this score while an edge between a cereal and legume crop would score more highly. Adding a field margin to an existing field would score higher still, but rather less so than approaches such as agroforestry or silvopasture. A measure of edge density could also be weighted to account for specific interfaces of habitats with disproportionate effects on specific ESs; for example, a flowering field margin next to an insect-pollinated crop (Gillespie *et al.*, 2022) is likely to be particularly beneficial, and would receive a higher score. Edge density will tend to correlate strongly with measures of connectivity and the relative benefit of an increase in edge density should be interpreted in this context; where new landscape features are created this should be done in a way that connects existing non-crop habitats.

(4) **Resource ratio hypothesis: balance resource ratios better and improve nutrient-use efficiency of system**

The abundance of ES providers will be positively related to the availability of resources (see Section II.1), whereas species richness and community composition at the farm scale will be determined by niche partitioning and spatial heterogeneity in resource provision (see Section II.2). Within a habitat type, however, the temporal demand for resources and ability to capture them will also differ between species. This concept can explain species coexistence through finer scale niche partitioning within habitat types: species can coexist whenever there are multiple limiting resources and species vary in their demand for different resources (Braakhekke & Hooftman, 1999; Harpole & Tilman, 2007; Harpole *et al.*, 2016). The increase in the availability of any one resource to the extent that it becomes non-limiting shifts the limitation towards a lower number of resources. In the most extreme case, all species will be limited by a single identical resource, and the potential niche dimensionality will be severely reduced, leading to species loss and increased dominance. Conversely, species richness will be higher when resources are available in a balanced manner regarding the needs of the different species in a community (Cardinale *et al.*, 2009).

One of the characteristics of modern farming systems is the reliance on mineral fertilisers to provide crops with the nutrients required for maximum growth and yield. In this sense, cropped fields (and to a lesser extent, pastures) are deliberately managed to ensure these resources are non-limiting, leading to an imbalance of soil nutrients with a particular surfeit of nitrogen and phosphorus. This has had negative unintended consequences for agronomic and environmental sustainability. Firstly, the imbalance of nutrients has selected for a few dominant, nitrophilous, weed species (Storkey *et al.*, 2021) that tend to be particularly competitive for light, which along with water, is the main resource limiting growth in highly fertilised crops. Increasing fertiliser use has been identified as one of the main drivers of declining weed diversity in Europe (Storkey *et al.*, 2012) and it has been hypothesised that a more even ratio of below-ground resources would promote a less-competitive weed community (Smith, Mortensen & Ryan, 2010). The use of inorganic fertilisers has also been shown to weaken plant–microbe networks in soil, compromising the capacity for nutrient cycling (Huang *et al*., 2019), which particularly increases crop vulnerability to nutrient deficiency under drought conditions (Bowles *et al*., 2022). Finally, beyond the boundaries of the production system, excess nitrogen and phosphorus in the environment leads to declining water and air quality (through nitrous oxide emissions) and eutrophication of semi-natural habitats with impacts on biodiversity. It would, therefore, be beneficial to improve the balance of resources (and, by implication, nutrient-use efficiency) at the farm scale through reducing the requirement for inorganic fertilisers.

This could be achieved by growing more diverse crop rotations that differ in their resource requirements, particularly the increased use of legumes that do not require nitrogen inputs (MacLaren *et al*., 2022) and the integration of short-term leys into arable rotations (Austen *et al*., 2022). These interventions would be captured in a metric of crop functional diversity (Section II.2). In addition, alternative sources of nutrients to inorganic fertilisers could be used including green compost, farmyard manure and anaerobic digestate. These organic inputs have a more balanced stoichiometry than inorganic fertilisers, which may reduce competition between weeds and crops (Ryan *et al*., 2009), and they also bring additional benefits for soil function by building soil carbon that supports a microbial community with more efficient nutrient cycling (Albano *et al*., 2023; Neal *et al*., 2023; Rayne & Aula, 2020). Manure could be applied directly or through the integration of livestock into arable systems (MacLaren *et al*., 2019).

One approach to benchmarking farms in terms of the balance of their nutrient inputs is to include a measure of the different forms of nutrients such as nitrogen in an index of system diversity (MacLaren *et al*., 2019; Tamburini *et al*., 2020). However, a recent meta-analysis of the interaction of so-called 'Ecological Intensification' practices to support yield concluded that the benefits of applied manures or composts or additional biological nitrogen fixation by legumes acted in a largely substitutive way in terms of their interaction with inorganic fertiliser additions (MacLaren *et al*., 2022). Benefits of the approaches described above in terms of their contribution to sustainability are only realised, therefore, if inorganic fertiliser use is also reduced. We therefore propose the simple metric of 'nutrient use efficiency' from fertilisers (NUE, nutrient offtake in crops/input of nutrient as inorganic fertilisers) at the farm scale as a useful system property for capturing the impact of these changes in practice and likely effects on resource ratios. NUE can be calculated from overall fertiliser inputs and crop yields (using estimated or measured nitrogen and phosphorus contents). Practices such as addition of organic amendments, integration of livestock and increased use of legumes and other diverse crops have the potential to reduce inorganic fertiliser use while maintaining yields (Albano *et al*., 2023), and so would be expected to improve 'fertiliser yields' as well as providing the additional benefits of building soil carbon and reducing the environmental impact associated with the manufacture of inorganic fertilisers. Scaling by absolute yield (*sensu* Pittelkow *et al*., 2013) would avoid the potential danger of targeting improved NUE leading to systems with low productivity.

(5) Successional dynamics: avoid extremes of disturbance to balance productivity with function of later successional communities

It is well known in ecology that recently disturbed (early successional) ecosystems are 'leaky' in terms of nutrient cycling, whereas mid- to late-successional ecosystems have much tighter cycles of nutrient turnover (Chapin, Matson & Vitousek, 2011). The tightening of nutrient cycling is driven by later-stage, longer-lived species tending to possess traits that enable more efficient resource use at lower fertility levels than ruderal species (Raevel, Violle & Munoz, 2012; Reich, 2014) – the resource acquisition/conservation continuum. This is true of soil microbial communities as well as plants, and recent research has indicated that microorganisms receiving long-term carbon inputs in the absence of disturbance modify the physical and chemical structure of the soil to retain water and nutrients better (Neal *et al*., 2020). In grassland, fungal communities and their interactions with plant species may also take decades to establish after disturbance (Seaton *et al*., 2022). Most farmland, particularly that dominated by arable crops, is deliberately kept in an early successional state through frequent disturbance (using soil tillage or herbicides to remove extant biomass) so that regular flushes of resources are made available to, predominantly annual, crops (Smith, 2015). Along with high fertiliser inputs, this results in modern arable farming systems being dominated by extreme ruderal habitats characterised by frequent disturbance and high soil fertility. Nutrients are especially vulnerable to losses in these systems *via* the processes of runoff and erosion from bare ground combined with nitrification, leaching and decomposition of organic inputs with high rates of nutrient turnover (Dungait *et al*., 2012).

From a community ecology perspective, as well as tillage, the application of pesticides can also be seen as a 'disturbance event' further selecting non-crop species with extreme ruderal strategies of short life cycles, high fecundity and wide dispersal that enable populations to buffer localised, frequent disturbance (MacLaren *et al*., 2020). These tend to be generalist species with potentially high population growth rates and the propensity to become crop pests. This can be observed in weed floras that are now characterised by a small number of dominant ruderal species (Bourgeois *et al*., 2019) with dramatic declines in species less well adapted to buffer disturbance events because of low fecundity and/or transient seedbanks (Storkey *et al*., 2012). The dominance of fewer pest species in turn leads to a reliance on a smaller number of pesticide active ingredients and the evolution of resistance (Hawkins *et al*., 2019). Agricultural intensification has also selected for a depleted community of pollinators that is now dominated by generalists that are adapted to modern farmed landscapes with the loss of specialist species and potentially system resilience (Redhead *et al.*, 2018). It would be beneficial, both agronomically and environmentally, therefore, to reduce the selection pressure for extreme ruderal life-history strategies and broaden the ecological niche, increasing the functional diversity of farmland biodiversity.

As a first principle, practices should be followed that avoid large areas of land being in a 'pre-successional' state, i.e. bare ground, which is most vulnerable to resource loss and colonisation by weeds. Cover crops and catch crops can be employed in seasons when cash crops are not grown (Constantin *et al.*, 2010), and crop residues can be retained as physical protection as well as a carbon resource for microbes to immobilise nitrogen (Vogeler, Boldt & Taube, 2022). Secondly, practices that integrate perennial, later successional, vegetation into farmed landscapes that are currently dominated by annual crops, will move farms along the successional gradient. This can be done by integrating semi-natural habitats into the farm landscape, such as the use of riparian buffer strips or in-field strips of perennial vegetation (including 'beetle banks'; MacLeod *et al.*, 2004), using deep-rooted plants that can capture nutrients as water moves through the soil and input carbon directly into the soil from root matter, whilst also providing habitat for pollinators and predators of crop pests (Schulte *et al.*, 2017; MacLeod *et al.*, 2004). Agroforestry or silvopasture offers similar benefits (Pavlidis & Tsihrintzis, 2018; Boinot *et al.*, 2019) and all these interventions would also increase resource provision, habitat heterogeneity and edge density (Sections II.1, II.2 and II.3). Perennial forbs can be sown, potentially in addition to conventional ryegrass cultivars or other grass mixes in 'herbal leys', providing multiple environmental benefits (carbon storage, reduced emissions, resources for pollinators) as well as the potential for improved efficiency of livestock production (Jordon *et al.*, 2022). These leys may be used within arable rotations or within high-intensity grassland management systems (e.g. dairy), although the use of short-term leys for grassland cultivation is associated with creation of bare ground and associated issues and new systems approaches may be needed to replace current intensive systems (Delaby *et al.*, 2020). Finally, where annual crops are grown, intensive, inversion ploughing can be replaced with minimal tillage or direct drilling with benefits for soil health and below-ground functional biodiversity (Cooper *et al.*, 2021; Stroud, 2019).

One option for evaluating a farm in terms of its position along a gradient of vegetation successional dynamics would be to record directly the intensity and frequency of disturbance events (tillage, pesticide application, grazing or mechanical removal of biomass). Examples include the Treatment Frequency Index (TFI) for pesticides, where a single application of an active ingredient at field rate has a TFI of 1 and subsequent pesticide applications increase TFI additively, or a proxy such as number of herbicide application days (Hicks *et al.*, 2018). However, obtaining these detailed management data is often difficult and an alternative approach would be to assess the emergent landscape and habitat characteristics as an indirect measure of the frequency and intensity of disturbance. This could be done using remote-sensing data; for example, a combination of mean Normalised Difference Vegetation Index (NDVI) over a growing season and the intra-annual coefficient of variation has been proposed as a useful index of ESs and would capture periods of bare ground and disturbance events (Paruelo *et al.*, 2016). Secondly, information on crop and non-crop habitat, of the type used to quantify resource provision and habitat diversity (see Sections II.1 and II.2), could be interpreted in terms of life-history traits; for example, by weighting plant communities by their Raunkier life form to quantify proportions of habitat representing different successional stages.

The best environmental outcomes would be expected where disturbance is minimal, localised and/or irregular, to avoid creating strong selection pressure for problematic species and to avoid repeated instances of specific environmental impacts. However, some degree of disturbance in agriculture will always be necessary to: (*i*) maintain grazing for livestock production and interrupt successional processes to prevent annual cropping systems becoming dominated by perennial weeds, and also on arable to (*ii*) disrupt the life cycles of pests, weeds and diseases adapted to low disturbance and (*iii*) provide habitat for biodiversity adapted to frequently disturbed habitats (such as farmland birds and threatened arable weeds) (Butler *et al.*, 2009). When seeking to manage successional gradients at the farm scale for agronomic sustainability, a balance between the benefits and costs of disturbance could be achieved by integrating crops and/or pastures with different longevities, and by retaining patches of undisturbed non-crop habitat (which also supports ES providers).

III. DISCUSSION

(1) Gradients of sustainability

In meeting the challenge of how to quantify farm sustainability, rather than starting with the selection of indicators or metrics, we have taken the novel approach of beginning with a review of the current state of knowledge on the ecological processes that determine the regulating and supporting services that underpin food and ES production on farmland. Based on this fundamental understanding of the behaviour of the agroecosystem, properties of the farm were identified that are predicted to reflect the relative integrity and status of these ecological functions. Regardless of the farm system terminology that is used (Fig. 1), we propose the resulting list of farm-scale metrics, summarised below, that can be used as proxies for ESs to identify beneficial directions of travel, benchmark farms and monitor progress along gradients of sustainability (Fig. 6). These metrics are not prescriptive of the exact practices required to achieve them, and so offer a shared framework of common goals to bring together

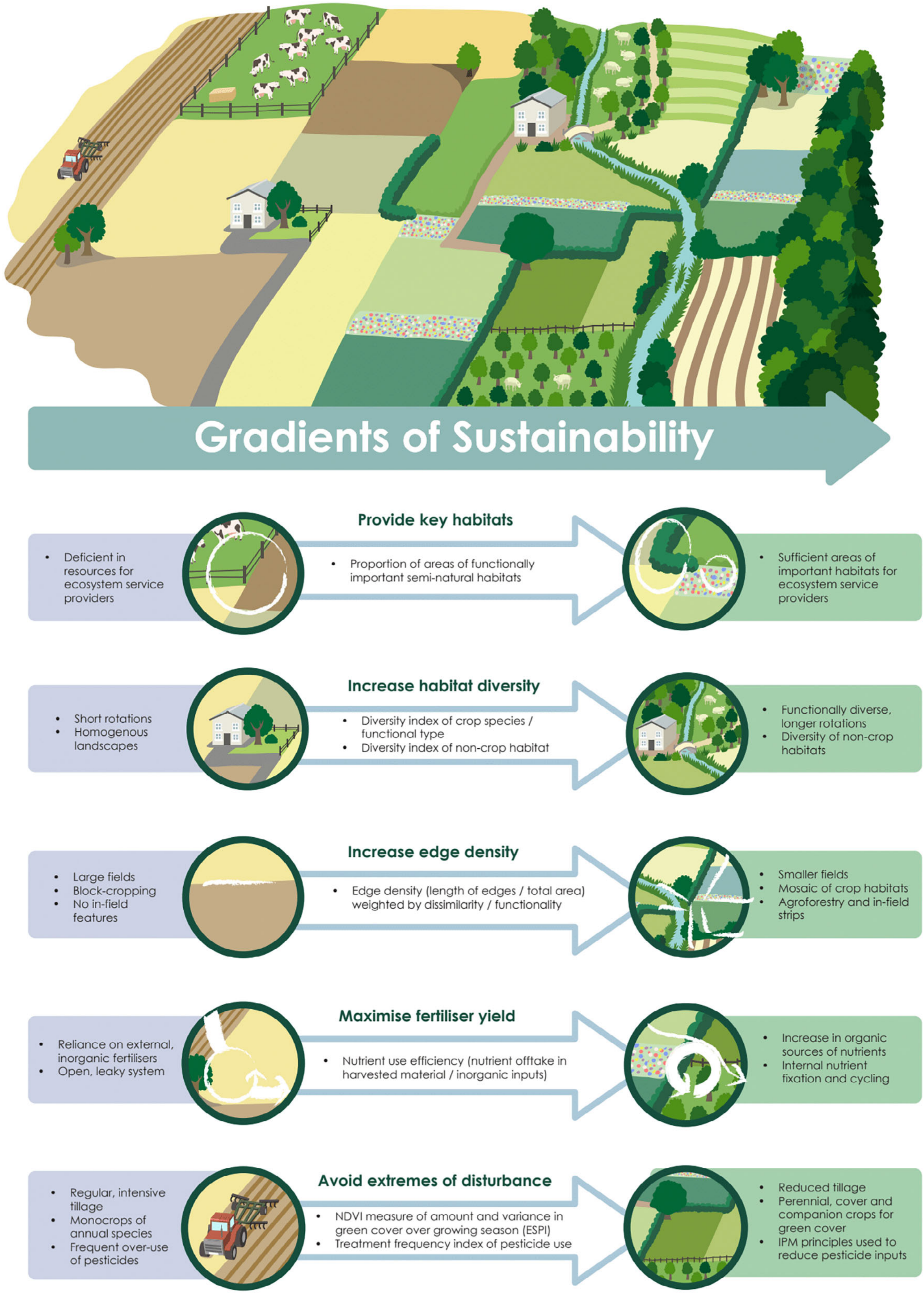

Fig. 6. Gradients of farm sustainability quantified using metrics informed by ecological theory and guiding principles for system change identified in Fig. 2. ESPI, ecosystem service preservation index; IPM, integrated pest management; NDVI, normalised difference vegetation index. Illustration by www.mair.perkins.co.uk.

different perspectives in sustainable agriculture. There is space for different methods and different priorities (for example, avoiding chemical inputs in organic farming or enhancing soil health in regenerative agriculture) while working towards increasing these metrics of overall ecological functionality of farming systems. They start from where most of our systems now are but aim for much more integrated agroecological systems where crop and non-crop habitats intermingle with diverse cropping regimes to enhance both food and ES production.

By focusing on agronomic and environmental sustainability, we have deliberately prioritised provisioning services delivered by farmland to inform the fundamental re-building of agroecosystem integrity at the farm scale such that the delivery of regulating and supporting services becomes an intrinsic property of the system. This gives rise to metrics that measure 'win–win' scenarios, where farm productivity benefits from restoring ecosystem integrity (Fig. 2), which will likewise deliver benefits to the environment and society beyond farm boundaries. In suggesting metrics, below, we have taken into account the criteria identified by (de Olde *et al.*, 2017) and taken the pragmatic approach of using earth observation data or farm records avoiding the need for the collection of additional empirical data:

(*a*) *Proportion of areas of functionally important habitat*

Using high-resolution satellite data or on-farm habitat surveys, record areas of functionally important habitats (for the ESs discussed here, areas of pollen and nectar provision, tussocky grass margins and hedgerows). Express either as separate proportions of total farm area or combine using sum of reciprocals. Option to weight further by habitat quality.

(*b*) *Diversity indices of crop and non-crop habitats*

Using high-resolution satellite data, farm crop maps and/or on-farm surveys, record areas of different crop species and non-crop habitat types. Express using indices of diversity or evenness calculated on an area basis (for example, Simpson, Shannon or Pielou index; Magurran, 2003). For crops, this could be calculated based on genotypic or functional diversity. Non-crop habitats could be assessed using established habitat classification schemes (Butcher *et al.*, 2020) with the option of creating further subcategories relevant to the ES of interest.

(*c*) *Edge density*

Using high-resolution satellite data or farm maps of crops and semi-natural habitat, measure the length of interface between differing habitats and divide by total farm area. This can include crops adjacent to semi-natural habitats or interfaces between two different crop types. There is the option of weighting edges by their potential contribution to ecosystem function, for example, upweighting an interface of an insect-pollinated crop with a flower-rich habitat.

(*d*) *Nutrient-use efficiency*

Using farm records of inorganic fertiliser inputs (expressed as total kg of nitrogen or phosphorus applied on the whole farm in a single cropping season or averaged over several years) and nutrient offtake, calculate nutrient-use efficiency as offtake/inputs at the level of the farm. Nitrogen offtake can be calculated from yields and nitrogen concentration of harvested material. Where nitrogen concentration is not known, it can be estimated based on crop type.

(*e*) *Ecosystem service provision index (ESPI)*

Based on analysis of satellite data, two attributes of the seasonal dynamics of the NDVI can be combined to give a measure of overall productivity and variation in time and space of green cover (a proxy for disturbance) (Paruelo *et al.*, 2016). The annual mean NDVI (NDVImean), an indicator of light interception and hence of productivity, and the intra-annual coefficient of variation of the NDVI (NDVICV), a descriptor of seasonality, can be combined into an ES provision index [(ESPI) = NDVImean $\times$ 1 − NDVICV] calculated at the farm scale. If remote-sensing data are not available, proportions of different crops and non-crop habitats, and information on their life histories, could be used to estimate disturbance frequencies.

(*f*) *Treatment frequency index (TFI)*

Using farm data on frequency and rates of pesticide inputs, the TFI represents a proxy for intensity of disturbance from pesticide use and selection pressure for the evolution of pesticide resistance. The TFI is calculated by dividing the total amounts of active ingredients used in each crop by the standard doses assigned to each use of the active ingredient and can be further refined to include the environmental toxicity of different active ingredients (Kudsk, Jorgensen & Orum, 2018). The TFI calculated for each crop can be summed across the whole farm and expressed on a per hectare basis.

The list of six metrics above is founded on a consensus of knowledge from the ecological literature and as such is not contentious, but our focus on regulating and supporting ESs delivered to production systems by biological communities means that our list is inevitably incomplete if outcomes other than sustainable production and agrochemical use are considered. For example, we have not referenced the extensive literature on metrics of soil health that may have a primary focus on alternative outcomes that are external to the production system, such as climate change mitigation (Chabbi *et al.*, 2017). However, our approach of using unifying scientific theory to interpret the impact of alternative approaches to farm management and identify outcomes and metrics has the potential to be applied more widely in other disciplines to help build a framework for quantifying farm sustainability that has broad acceptance across the academic, policy and practitioner communities.

(2) Implementation

Basing our rationale for the selection of metrics on established ecological theory gives confidence that they are effective proxies for agroecosystem function. They can, therefore, be used in the first instance to monitor and inform the 'direction of travel'; an improvement in one or more of the metrics is assumed to have a net beneficial effect for one or more ES (Fig. 2). As the metrics can also be derived from remote-sensing data or existing farm management records, they also provide a basis for baselining and benchmarking farms along the gradients of sustainability (Fig. 6) without the need for additional surveys. Comparing farms at the regional or catchment scale would account for potential large-scale, confounding effects including variation in regional species pools. It would also allow the metrics to be scaled beyond the individual farm to inform management of farm clusters or local landscapes.

However, the implementation of our framework would benefit greatly from further quantitative analysis of the relationships between directly measured data on ESs (or ES providers) and the farm-scale metrics we have identified. These analyses could be done retrospectively on published data or new data derived from traditional ecological surveys or sampling methodologies. In the future, it is likely that new technologies (computer vision, acoustic monitoring, radar, and molecular methods) will transform our capability to monitor biodiversity at the farm scale (van Klink *et al.*, 2022). Using these data to challenge our framework will be valuable for: (*i*) providing additional evidence to support the assumption that the metrics are effective proxies for ESs and for the underlying theories; (*ii*) calibrating and quantifying the shape of the relationship between metrics and ESs (identifying nonlinearity and thresholds); and (*iii*) contextualising the relationships between metrics and ESs at the regional scale (thereby capturing effects of regional species pools and landscape structure).

IV. CONCLUSIONS

(1) The concept of farm sustainability can be difficult to evaluate objectively because it depends on the choice of system outcomes used in an assessment framework and their inherent trade-offs and synergies. A 'results-based' framework that measures outcomes (including ESs) directly also has the challenge of developing robust methodologies for the quantification of outcomes that may respond to processes operating over different spatial and temporal scales. Using an alternative, 'practice-based' approach to benchmark and monitor sustainability, while more tractable, is also challenging because the impact of any one practice on an outcome will depend on the local system and environmental context. Evidence on the relative benefits of alternative practices, and potential trade-offs elsewhere in the system, is also often lacking.

(2) With the aim of developing a scalable and practical framework for benchmarking in terms of agronomic and environmental sustainability, we took the pragmatic approach of focussing on the 'middle ground' between these two approaches. Guided by the consensus of evidence in the literature and fundamental ecological theory, we identified properties of farms that are both responsive to land-use and management change and determine ES delivery and downstream outcomes – i.e. proxies for agroecosystem function. Focussing on this interface between practice and outcomes provides a basis for decision making ('how will a possible management change impact one or more system properties?') and monitoring progress towards more sustainable systems.

(3) The resulting gradients of sustainability we identified could be used to inform and promote incremental change at scale that does not presume categorical system change (e.g. from so-called 'conventional' to 'regenerative'). In so doing we acknowledge the challenge of system categorisation and that farms sit along a continuum defined by different measures of sustainability (and are at different starting points). A given farm can, therefore, be assessed against the six metrics to identify areas for improvement based on a comparison with farms in a similar region or landscape. For example, a farm may score highly against the criteria of 'provision of key habitats' but these may not be well integrated within the cropped areas of the farm (measured by 'edge density'), limiting the provision of ESs to support crop production.

(4) The implementation of our framework is predicated on the integration of data at the farm scale on land use (habitat maps) and management (fertiliser and pesticide use) and the full value of a set of metrics for an individual farm will only be realised in the context of equivalent data from multiple farms in the same landscape or region. As well as validating the relationships between system metrics and ESs, therefore, there is also a parallel need to develop scalable methodologies for collating these data based on Earth observation, digital farm maps and farm management records.

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