Agroecology – a Rapid Evidence Review

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The CCC commissioned The University of Aberdeen to assess the role of agroecological farming in the UK in the transition to Net Zero. The review reflects the evidence at the time of writing and was commissioned to inform the CCC's developing work programme for land and agriculture. As such it does not necessarily represent the view of The Committee.

1 Executive Summary

1.1 Aims

Agroecology is a less intensive farming system than conventional agriculture, based on utilising the natural interactions between plants, animals, humans and the environment to support sustainable food production, whilst restoring ecosystem services and building resilience to climate change. In particular, Agroecology targets the diversification in agricultural approaches, offering great potential to support the sustainability of the transition of agriculture towards net zero. **This report reviews the scientific evidence on the environmental impact of agricultural practices that are used in agroecology that are proposed to enhance productivity and resilience of agroecosystems.**

To introduce the principle of agroecology in the net zero pathway, it is important to specify the characteristics that identify different agricultural practices as agroecological. For nineteen agricultural farm practices (AEFPs) in arable and livestock systems, **we review the scientific evidence on their potential to address six agroecological principles at farm level (recycling, input reduction, soil health, animal health, biodiversity and synergy), as well as their impact on greenhouse gas emissions (GHG) emissions, vegetation and soil carbon stocks, and changes to yields (Figure 1).**

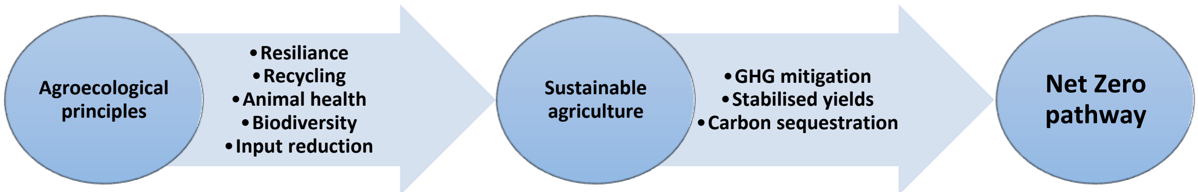


Figure 1: The reduction of agricultural emissions towards net zero will need the implementation of agroecological farm practices (AEFPs) intended to enhance the sustainability of the changes required in agriculture. This report reviews the level of sustainable transition achievable by different AEFPs targeting low carbon farming, higher agricultural productivity, agroforestry and hedges.

Each AEFP can be classified along a spectrum and designated as more or less agroecological depending on the extent to which different agroecological principles are applied. **In this context, we summarise the results of the literature review to provide the total number of benefits (positive impact) and**

trade-offs (negative impact) that each AEFP can generate at farm-level. AEFP that score high in total number of benefits can be seen as more effective for transitioning agriculture toward a sustainable system *via* agroecology.

We highlight the opportunity to introduce agroecology in the Balanced Net Zero Pathway Scenario used by the Climate Change Committee in their Sixth Carbon Budget Advice (CCC, 2020a). In particular, we discuss how agroecology can support the sustainability of the agricultural changes needed in the transition towards net zero, outlining its environmental benefits and trade-offs, the knowledge gaps that need to be addressed, and the direction of future work.

1.2 Key Findings

- Due to still-limited scientific knowledge or low practical on-farm experience the majority of the AEFPs reviewed here currently have only a low or medium implementation level in UK agriculture today.
- AEFPs with a relatively low level of integration such as: intercropping, multispecies leys, ley-arable rotations, perennial cereals and vegetated strips can generate high level of agroecological transition. This means that these practices have the potential to generate win-win situations from reduction in chemical inputs and land-use change, the improvement of soil properties, livestock diets, species richness, resource management, and ecosystem services such as water quality, air quality, and resilience at farm level. However, the time delay before ecological benefits are realised from these AEFPs means that significant investment, training and re-design of the farm business are required.
- A number of AEFPs that score high in agroecological principles such as rotational cropping, cover crops, legume crops, and permanent pastures are already well integrated in UK's agriculture, indicating that there are opportunities to speed the agroecological transition with relatively low investments. Of the nineteen AEFPs, fifteen showed positive impact on soil carbon stocks, with application of organic manures, conversion to permanent pasture and adopting multifunctional land use systems achieving increases in SOC in excess of 30% compared to conventional practice.
- No tillage, reduced tillage, pasture cropping, retention of straw, cover crops, silvoarable and the conversion of temporary grassland to permanent pasture can reduce productivity at farm level, although yield losses are typically less than 10%.
- We found that well-studied agroecological practices such as no tillage, retention of straw, organic manure and cover crops can increase GHG emissions, resulting in some cases in more than doubled emissions of nitrous oxide. However, significant knowledge gaps remain in understanding the net impact on GHG emissions from adopting the majority of AEFPs reviewed here, particularly regarding changes in methane emissions from adoption of AEFPs in ruminant livestock systems.
- In the Balanced Net Zero Pathway Scenario, agroecology could be introduced through practices such as: low carbon farming, options to release agricultural land, as well as use of agroforestry and hedges. In particular, agroecological approaches could be implemented in the measures targeting the improvement of livestock diets and health, soil measures, and higher agricultural productivity.

2 Contents

2 Introduction

Agriculture is the fourth largest contributor to total UK greenhouse gas (GHG) emissions (54.6 million tonnes of CO_2 equivalents ($MtCO_2e$), 11% of total), after energy, transport, business and residential construction sectors. Of these emissions, 53% is derived from animals, predominantly from enteric fermentation in ruminants (i.e. cattle, goats, sheep), 21% from the management of soils, particularly application of fertilisers, 16% from manure and waste management, and 8% from mobile and stationary agricultural machinery (CCC, 2020b).

To place the agriculture, forestry and other land use (AFOLU) sectors on a pathway to Net Zero by 2045/2050, the Committee on Climate Change (CCC) advocated a number of changes in the use of land across the UK that could deliver a 64% reduction of GHG emissions from AFOLU (CCC, 2020b). In particular, in the agricultural sector these changes entail an improvement of the agricultural productivity to facilitate land release across six areas of opportunity: i) the take-up of smart technologies and precision farming to reduce non-CO_2 emissions from soils, ii) land management for carbon sequestration, iii) new solutions in the areas of animal husbandry, plant and animal breeding and livestock welfare, iv) novel and alternative feeds, v) reduction of agricultural waste, and vi) switching away from fossil fuel use in agricultural machinery to low-carbon alternatives.

The reduction of agricultural emissions towards net zero will also require the implementation of agroecological principles intended to enhance the sustainability of the required changes. In this context, agroecology can support transitions to diversified and resilient food systems, while also addressing the need for socially equitable food systems where people can exercise choice over what they eat and how and where it is produced (Notenbaert et al., 2021). Wezel et al. (2020) articulated agroecology as a framework of 13 principles that include: recycling, input reduction, soil health, animal health, biodiversity, synergy, economic diversification, co-creation of knowledge, social values and diets, fairness, connectivity, land and natural resource governance and participation. However, in order to introduce the principle of agroecology in net zero strategies it is important to specify the characteristics that identify innovative agricultural practices as agroecological. An innovative practice can be something completely new, but also a practice based on old agricultural principles or techniques that is adapted to modern agriculture (Uphoff 2002), thus creating a novelty for application.

Examples of agroecological practices are already discussed in the literature, e.g. cover crops, green manure, intercropping, agroforestry, biological control, resource and biodiversity conservation practices, or livestock integration (Wezel et al., 2014). However, there are no clear boundaries between what is agroecological and what is not (Wezel et al., 2020). On the contrary, agricultural practices can be classified along a spectrum and designated as more or less agroecological, depending on the extent to which agroecological principles are applied.

Opportunities for designing sustainable agroecosystems imply different levels of modification within the farms, either at field scale, at the cropping/farming system scale, or a landscape scale. In their review, Wezel et al. (2014) articulated the advantages and constraints of different agroecological farm practices (AEFP) according to the analytical framework of Hill and MacRae (1995). Their framework describes the sustainable transition of agriculture based on three non-linear stages of intervention: i) efficiency increase, ii) substitution practices, and iii) farm re-design. EI refers to practices that reduce input consumption within the farm boundaries (e.g. water, pesticides, and fertilisers) and is characterised by a low level of requirements for change and knowledge of the farmers. In general, SP corresponds to the substitution of an input or a practice (e.g. replacing chemical pesticides by natural pesticides), and may require low or medium level of changes. FR is a more radical change for the farmers and entails a change of crop system, herd characteristics, or even farming system. For example, changing crop cultivar can be relatively easy to implement, but the modification of crop rotation systems or the development of agroforestry requires redesign and reorganisation of the practices, new machinery, new buildings, and the need for the farmer to enhance their knowledge. Although, EI and SP tend to be additive and incremental within current production systems, they are not sufficient for maximizing co-benefits of smart agricultural and beneficial environmental outcomes without FR (Pretty, 2018). However, FR presents agricultural challenges generated by medium- and long-term economic trade-offs that explain why some AEFPs are not yet currently widely applied in UK agriculture.

Here, we review a number of agricultural practices, in relation to the first six principles of agroecology:

1. *Recycling*. Preferentially use local renewable resources and resource cycles of nutrients and biomass within the farm boundaries.
2. *Input reduction*. Reduce or eliminate dependency on external inputs increasing self-sufficiency through the use of alternative species mixtures in grassland and arable rotations, and enhancing home-grown sources such as protein from nitrogen (N) fixing legumes.
3. *Soil health*. Secure and enhance soil health and functioning for improved plant growth, particularly by managing organic matter and enhancing soil biological activity.
4. *Animal health*. Development of new solutions in the areas of animal husbandry, plant and animal breeding and livestock welfare.
5. *Biodiversity*. Maintain and enhance the overall agroecosystem biodiversity of species, functional diversity and genetic resources in time and space at field, farm and landscape scales.
6. *Synergy*. Enhance positive ecological interaction, synergy, integration and complementarity amongst the elements of agroecosystems (animals, crops, trees, soil and water).

In particular, for each innovative agricultural practice in arable and livestock systems we review quantitative and qualitative evidence on the potential impact on GHG emissions, yields, and vegetation and soil carbon stocks. In addition, we review their advantages and constraints, and potential for further adoption.

3 Definition of agroecological farming practices

Here, we review a set of agricultural practices that can support the implementation of agroecology in UK agricultural systems. We group these in to six categories based on their characteristics and aims:

1. Reduce soil disturbance
2. Organic inputs
3. Diverse crop rotations
4. Multifunctional land use
5. Pasture productivity
6. Livestock extensification

Table 1 outlines the nineteen AEFPs reviewed in this report and their potential compatibility and incompatibility. Some of these practices have already been applied to varying degrees in different regions of the UK for decades (e.g. conservation tillage, cover crops, or legume crops), while others have been more recently developed and still have a limited scale of application (e.g. perennial cereals, pasture cropping, silvoarable). In developing agroecological practices, the question of landscape multifunctionality is inevitable (Wezel et al., 2014), and it will require AEFPs such as agroforestry (silvopasture and silvoarable), multispecies leys or intercropping to be reintegrated into cropping systems to enhance their resilience to climate change (Malézieux et al., 2009; Tilman et al., 2006).

Table 1: Agroecological farm practice compatibility / incompatibility matrix. Note that liming does not constitute an agroecological intervention. In croplands and grasslands this practice can contribute to offsetting the negative impact of conventional agriculture. Therefore, we include liming in this review as a transitional practice to support longer term agroecological outcomes.

	No-till	Minimum tillage	Retention of straw	Organic manure	Cover crop	Intercropping	Legume crops	Ley-arable	Pasture cropping	Perennial cereal	Vegetated strips	Silvoarable	Silvopasture	Rotational grazing	Extensive grazing	Multispecies leys	Temporary leys to permanent pasture	Grass-based livestock diets	Liming
No-till		X	✓	a	b	✓	✓	b	R	✓	✓	✓	na	na	na	b	na	na	✓
Minimum tillage			✓	c	b	✓	✓	b	X	✓	✓	✓	na	na	na	b	X	na	✓
Retention of straw				✓	✓	✓	✓	na	X	✓	✓	✓	na	na	na	na	na	na	✓
Organic manure					✓	✓	✓	✓	a	a	✓	✓	✓	✓	d	✓	d	na	✓
Cover crop						✓	✓	✓	X	X	✓	✓	na	na	na	na	na	na	✓
Intercropping							R	✓	X	ü	✓	✓	na	na	na	na	na	na	✓
Legume crops								✓	✓	X	✓	✓	na	na	na	na	na	na	✓
Ley-arable									X	✓	✓	✓ e	✓ e	✓ f	✓ f	✓ f	X	✓	✓
Pasture cropping										X	✓	✓ e	✓ e	✓ f	✓ f	✓ f	✓	na	✓
Perennial cereal											✓	✓	na	na	na	na	na	na	✓
Vegetated strips												✓	✓	✓	✓	✓	✓	na	✓
Silvoarable													na	na	na	na	na	na	✓
Silvopasture														✓	✓	✓	✓	✓	✓
Rotational grazing															✓	✓	✓	✓	✓
Extensive grazing																✓	✓	✓	✓
Multispecies leys																	g	✓	✓
Temporary leys to permanent pasture																		✓	✓
Grass-based livestock diets																			na
Liming																			

✓	Compatible
✓	Compatible with caveats, see footnotes corresponding to letter
x	Incompatble
R	Required
na	Not applicable

(a) Solid manure typically incorporated into cropland by ploughing. If solid manure applied in no-till system, would require spreading on stubble in autumn so could naturally incorporate into soil before fields sown in spring, or spread lightly once crop growing; to prevent crop establishment being hindered. No issues with liquid manure (i.e. slurry).

(b) Would require herbicide use to terminate cover crop/ley-phase/previous ley if direct drilling (some possibility to mechanically terminate cover crops, but use not widespread).

(c) Would require application to be sufficiently light to be able to be incorporated via reduced cultivation method.

(d) If organic manure applied to pasture, livestock often have to be removed for period after due to grass contamination (hence easier in rotational grazing system).

(e) Systems with integrated arable, pasture (ley) and trees referred to as agrosilvopasture.

(f) Possible to graze ley/pasture when crops not growing in rotation.

(g) *Can increase sward diversity to include legumes and forbs through 'overseeding' (i.e. direct drilling) approaches, but can be less successful than termination of ley via spraying/ploughing*

Therefore, the implementation of different AEFPs implies different degrees and scale of changes within the farming systems. Table 2 shows the degree of change (low, medium and high) and scale of implementation characterising each practice. In this context, the scales of implementation can be at field level, cropping system level, or landscape level when the provision of ecosystem services arise from diverse land-uses. This two-dimensional framework is applied to understand the implication of the changes for which distinct sustainable indicator measures are relevant, and the stage of sustainable transition achievable from each AEFP.

Table 2: Framework to classify agroecological farm practices (AEFP). Degree of change corresponds to the changes required to inplement AEFPs in conventional farms and obtained from the author's expert judgment. Scale of application corresponds to the spatial scale to which agroecological principles are locally applied. Transition stage corresponds to the stages of agroecologial transition achieved by the AEFP: i) efficiency increase (EI), ii) substitution practices (SP), and iii) farm re-design (FR).

Category	Practice	Degree of change	Scale of application	Transition Stage
Reduced soil disturbance	No-till and minimum tillage	Low	Field	EI
	Perennial cereal crops	Medium	Cropping system	SP-FR
	Pasture cropping	Medium	Cropping system	SP-FR
	Liming	Low	Field	EI
Organic inputs	Retention of straw	Low	Field	EI-SP
	Organic manure	Medium	Field	SP
Diverse crop rotations	Cover crop	Medium	Cropping system	SP
	Intercropping	High	Cropping system	FR
	Legume crops in rotations	Medium	Cropping system	SP
	Ley-arable	High	Cropping system	FR
Multifunctional land-use	Silvoarable	High	Cropping system, Landscape	FR
	Silvopasture	Medium	Cropping system, Landscape	FR
	Vegetated strips	Low	Field	FR
Pasture productivity	Rotational grazing	High	Livestock system	EI-FR
	Multispecies leys	Low	Field	EI-SP
Extensive livestock	Extensive grazing	High	Livestock system	FR
	Grass-based diet	Medium	Livestock system	SP
	From temporary leys to permanent pasture	Low	Field	SP

There are six practices with low degree of change, and seven practices that require medium degree of change. However, independently of the scale of application, more than 80% of the AEFPs entail substitution of conventional practices or re-design of agricultural activities within the farm. This means that their implementation may require a relatively high level of investment and change to drive an agroecological transition.

In the following sections we define each AEFP and highlight their agricultural aims. By contextualizing these outcomes within the agroecological framework outlined in Table 2, we aim to highlight where the best opportunities lie for the expansion of agroecology in UK agriculture.

3.1 Reduced soil disturbance

Reduced tillage intensity

Reduced tillage is an umbrella term encompassing many types of tillage and residue management systems that aim to achieve sustainable and profitable agriculture (Abdalla et al., 2013). Reduced tillage might include non-inversion tillage, eco-tillage, minimum tillage, mulch tillage, reduced tillage and no-tillage (no-till). In this review, in particular, we focus on the effect of reducing tillage intensity through no tillage or minimum tillage.

Conservation tillage systems are primarily based on reducing soil disturbance by restricting any land preparation activities to a shallow depth and eliminating soil inversion, while conserving and managing crop residues (Cunningham et al., 2004). Conservation tillage aims to leave at least 30% of the previous crop residues remaining on the soil surface, whereas conventional tillage leaves less than 15% (Gebhardt et al., 1985). In general, conservation tillage has been recommended for soil and water conservation, reduction in labour and energy costs as well as provision of many ecosystem services such as carbon sequestration and soil biodiversity conservation (Lal et al., 2007; Triplett & Dick, 2008).

Perennial cereal crops

Perennial cereal crops are a relatively new crop option for farmers (Hayes et al., 2012). Their development, *via* wide hybridisation of annual cereals with perennial wild relatives, represents a feasible opportunity to reduce the environmental degradation associated with annual crops, while providing a viable alternative in conventional agriculture (Jaikumar et al., 2012). Perennial wheat and perennial rye, in particular, are two promising candidates for cold-temperate regions, with the potential to be cultivated in marginal agricultural land. To date, a number of studies have shown that the perennial wheat grain crops grown on marginal land can have the same environmental benefits as perennial bioenergy crops (Bell et al., 2010; Glover et al., 2010). The extensive root system may encourage soil organic carbon (SOC) accumulation in both the topsoil and the subsoil and may positively influence microbial biomass and activity in these soil horizons (Duchene et al., 2020).

Pasture cropping

Pasture cropping is a no-till technique consisting of sowing annual crops into living perennial pastures during their dormant stage (Lawes et al., 2014; Luna et al., 2020; Millar & Badgery, 2009). This cropping system combines species with complementary growth periods to diversify the farming systems and improve overall land productivity. The most commonly accepted practice involves establishing summer active perennial pastures that are grazed up to the autumn, when winter cereals are directly drilled in the dormant sod (Descheemaeker et al., 2014; Lawes et al., 2014; Millar & Badgery, 2009). In the spring, as temperatures increase, the pasture resumes growth when the winter cereal is in its final growth stage.

Liming

Liming is a common practice in conventional agriculture to neutralise and control soil acidification. Soil acidification (i.e. low soil pH) is a natural process that can be accelerated by use of nitrogen fertilizers. Soil acidification reduces vegetation productivity by decreasing soil base status and nutrient availability, and increasing the solubility of metals such as aluminium, iron and manganese that can be toxic (Kunhikrishnan et al., 2016; Holland et al., 2018; Horan et al., 2018). The use of lime in soil can contribute to reducing soil acidification by increasing soil pH, improving soil physical condition, reducing aluminium toxicity, and increasing soil phosphorus (P) and magnesium (Mg) availability when needed (Tunney et al., 2010; Holland et al., 2018).

Although liming does not constitute an agroecological intervention, in croplands and grasslands this practice can contribute to offsetting the negative impact of conventional agriculture on soil health and crop yields. Therefore, we include liming in this review as it can support the implementation of other innovative practices, stabilizing productivity both in cropland and grassland systems (Abdalla et al., 2022).

3.2 Organic inputs

Organic inputs are a core management strategy to improve soil health and mitigate climate change (Lazcano et al., 2021). Farmyard manure (FYM), slurry, and plant residues (left at the soil surface or incorporated) are the main sources of organic amendments to the soil. Organic fertilizers can indirectly increase soil C storage by increasing net primary productivity and root litter and exudation, a mechanism, which has been recently found to contribute to most of the sequestered or stable C in soils (Ryals & Silver, 2013; Sokol & Bradford, 2019). In this review, we focus on the environmental impact of organic manure (i.e. farmyard manure, slurry, and compost) and retention of straw.

Retention of straw

The return of straw in the soil can potentially improve the nutrient status of agricultural soils (Lal, 2004), which in turn stimulates SOC sequestration due to increased crop rhizodeposition (Kuzyakov & Schneckenberger, 2004). Returning straw into soil does not only directly increase C input into the soil, but also influences soil physical and chemical properties (Liu et al., 2014). In particular, the cover of crop residues on the soil can protect the soil surface against insolation and erosive impacts of raindrops and wind. It buffers the soil surface from excessive compaction, surface sealing, and crusting while reducing the breakdown and dispersion of soil aggregates (Blanco-Canqui, 2009).

Organic manure

Organic manure consists of predominantly urine and faeces, but depending on the specific livestock management practices it may contain spilled animal feed, water, and soil (He et al., 2016). The use of organic manure as a fertilizer constitutes a direct input of C to the soil, which can be stabilized through physical, chemical, and biochemical mechanisms contributing to long-term storage of C in soils. In addition, fertility improvement through an effective management of these soil properties has the capability of optimizing crop yields (Diacono & Montemurro, 2011). Compost is a stabilized and sanitized product of composting, which is the biodegradation process of a mixture of organic substrates carried out both in aerobic conditions and solid state (Insam & de Bertoldi, 2007). During the composting process, the simple carbonaceous and nitrogenous compounds of the organic matter are transformed through the activity of microorganisms into more stable complex organic forms, which chemically resemble soil humic substances (Epstein, 1997). Regular addition of compost or manures to soil for long periods can enhance both soil C and N stocks, indicating a physical protection of this nutrient within macroaggregates (Sodhi et al., 2009; Whalen & Chang, 2002).

3.3 Diverse crop rotations

Cover crops

In agroecology, cover crops (also known as catch crops) are grown for ecological services (e.g., soil cover, nutrient capture, fertility improvement, weed suppression) rather than a harvestable product (Moore et al., 2006; Nie et al., 2008). Cover crops are plants mostly grown after a primary crop is harvested, in arable systems where only a single main crop is grown (IPCC, 2006). Cover cropping can comprise a single species or a mixture of species and can use annual, biennial or perennial vegetation. Cover crops can be killed by herbicides (though not in agro-ecology) or ploughed-in in winter or spring, or grazed, and incorporated in soils by tillage to prevent competition with the primary crop and promote mineralization of organic N (Dabney et al., 2011; Halde et al., 2014).

In general, farmers select specific types of cover crops based on their own requirements, influenced by biological, environmental, social, cultural and economic factors of the farming systems in which they operate (Snapp et al., 2005). There are four classes of cover crops: legumes (e.g. alfalfa, vetches and clover), non-legumes (spinach, canola and flax), grasses (e.g. ryegrass and barley) and brassicas (e.g.

radishes and turnips), which are typically characterised as legumes and non-legumes. Legume cover crops have the ability to fix N biologically and increase soil organic matter (SOM) content (Lüscher et al., 2014). They can be used as a green manure to improve soil nutrition for the subsequent primary crop.

Intercropping

Multiple cropping is an agroecological practice where two or more species are grown simultaneously in the same field for at least a part of the growing season (Gaba et al., 2015; Rodriguez et al., 2020). Multiple cropping systems can be classified on their species composition, design, and management (Lithourgidis et al., 2011; Malézieux et al., 2009; Poveda et al., 2008). Although the primary aim of multiple cropping systems is to provide crop yields, these can provide other key ecosystem services. These are mainly regulating services that may include pest and disease regulation, erosion control, climate regulation, and maintenance of soil fertility (Gaba et al., 2015). Plant interactions in intercrops may take various forms a) competition, b) complementarity, c) facilitation and d) compensation (Cardinale et al., 2002; Tilman, 1996). Cereal–legume intercropping, in particular, is a well-known example of a system based on complementary functioning that optimizes the use of N at field scale over a growing season. Yields has been reported to increase in intercropping due to the complementarity in use of N sources (Hauggaard-Nielsen et al., 2008), or to the facilitation of interplant N transfer and availability of P (Jensen, 1996); (Hinsinger, 2001); (Li et al., 2009). Ultimately, in systems with little or no N fertilizer, intercropping systems can be more productive than monoculture with improved cereal growth, grain yield, and grain quality (Mahmoud et al., 2022).

Legume crops included in rotation

Legumes have traditionally been used in cropping systems as part of crop rotations, and also intercropped with other crops (especially cereals – see above). Legumes are an important source of protein for feed and food (Voisin et al., 2013). These crops have the ability through symbiotic microbial associations to fix atmospheric N which is returned to the soil, leading to a reduction in N fertilization needs, not only for their own production but also for the following grain crop (Costa et al., 2020; Hauggaard-Nielsen et al., 2003). Having legumes intercropped with non-legumes (i.e. cereals) could be a way to promote yield stability (Raseduzzaman & Jensen, 2017).

Ley-arable rotations

Grassland–cropping rotations have been the basis of the so-called ley farming systems for centuries in many areas of the world (Lemaire et al., 2015). These systems, based on rotation of crops with grass pastures, can be effective in improving soil structure and fertility and disrupting pest and disease lifecycles. The reintroduction of grasslands within arable cropping systems has the capacity to achieve synergy between productivity and achieving other ecosystem services (Franzluebbers et al., 2011). Further details on these benefits are reported in Section 4 and 5.

3.4 Multifunctional land-use systems

We review three types of multifunctional land-use systems reported in the literature: silvoarable, silvopasture and vegetated strips.

Silvoarable and Silvopasture

Agroforestry is a collective name for diverse land-use systems integrating tree husbandry with livestock or arable cultivation (Lundgren, 1982). Agroforestry is subdivided into silvopastoral systems, grazed by livestock or used for fodder production (Mayerfeld et al., 2021), and silvoarable systems, in which crops are grown among trees (Mosquera-Losada et a., 2009). In some instances, particularly in temperate regions, trees are grown only at the edge of a field (such as hedgerows adjacent to arable land) are occasionally defined as agroforestry systems (Mosquera-Losada et a., 2009, Schoeneberger et al., 2017). For the purpose of this review, we have separated agroforestry from set-aside vegetation strips, as their implementation may provide different implications and level of commitment for farmers, in the transition toward a sustainable food system *via* agroecology.

Agroforestry practices can be considered as a strategy to alleviate some of the environmental trade-offs related to the conversion of forested land to agriculture (Shrestha et al., 2004); (Nair, 2011; Orefice et al., 2019). However, it is still debated whether it is more beneficial to integrate diverse land uses on the same piece of land or keep them separate, particularly in the context of food production and biodiversity conservation (Lusiana et al., 2012).

Temperate silvoarable systems have the potential to increase productivity compared with equivalent mono cropped land (Graves et al., 2010); (Gruenewald et al., 2007). Timber is typically the main tree product produced of silvoarable systems, although intercropping with fruit trees is widely practised in China (Chang et al., 2018) and its potential for a quick return on investment is encouraging uptake in the UK (Newman et al., 2018).

Vegetated strips

Vegetated strips are defined as any vegetated area set-aside from the main cropping regime within or around a field, and installed for the purposes of benefiting biodiversity, water and air quality, and yield (Van Vooren et al., 2018). Examples of such interventions include: hedgerows, field-edge plantings, buffer or riparian strips, beetle-banks and shelterbelts (Haddaway et al., 2018). We focused on three types of semi-natural vegetation elements: hedgerows, grass margins, and tree strips. Both hedgerows and grass strips have been reported to deliver a number of ecosystem services, for example carbon sequestration and N and P interception from water flows (Van Vooren et al., 2018).

Farmers are frequently encouraged to plant herbaceous flowers or flowering hedges along the edges of crop fields as a means of increasing seed source for farmland birds, addressing wild bee declines and attracting wild pollinators to crop fields (Lowe et al., 2021).

3.5 Pasture productivity

Rotational grazing

Rotational grazing is a generic term used for diverse grazing management approaches that subdivide the grazing area into any number of paddocks that are grazed sequentially using pre-determined grazing periods (Teague & Kreuter, 2020). Using this method, livestock are concentrated on a smaller area of the pasture for a few days then moved to another section of pasture. This movement allows the grazed paddock a rest period that permits forages to initiate regrowth, renew carbohydrate stores, and improve yield and persistency. In addition, rotational grazing can help farmers increase forage productivity, and control the timing and intensity of forage grazed by the livestock.

Multispecies leys

In cold-temperate regions, simple swards containing perennial ryegrass (*Lolium perenne* L.) constitute the primary source of forage for livestock pasture-based systems (Mccarthy et al., 2020). A key advantage of simple swards is that they can withstand intensive grazing, and respond well to fertile conditions and N inputs (Shalloo et al., 2011). However, restrictions on fertilizer N inputs have incentivised the agricultural sector to find more sustainable pasture production systems based on species from more than one functional group, and requiring less fertilizer N inputs (Mccarthy et al., 2020).

Multispecies swards containing perennial ryegrass and white clover (*Trifolium repens* L.), and herb species (i.e. forbs), in particular, have been found to produce yields comparable to simple swards with low N requirements (Finn et al., 2013; Grace et al., 2018). The plant diversity of these systems can enhance the positive interactions between plants for resource acquisition and mobilization of natural regulation (Malézieux et al., 2009). In particular, plant diversity can provide a range of ecosystem services based on the type (positive, neutral, or negative) and degree of plant–plant interactions and on the local environmental and management conditions (Tilman., 1999; Diaz et al., 2006).

The complementarity of multispecies swards helps in the stabilization in the supply of herbage across seasons, making such grazing systems very relevant in sheep faming that rely on spring and autumn herbage supplies during lambing and breeding, respectively.

3.6 Extensive livestock systems

Extensive grazing (reduced stocking density)

Extensive grazing is a practice in which livestock are grazed at low stocking densities, often on semi-natural grassland and shrublands (Byrnes et al., 2018). Reduction of stocking density (i.e. grazing livestock units per hectare) allows for the provision of adequate forage, which can lead to an improvement in individual animal and total herd productivity compared with overgrazing (Hristov, Ott, et al., 2013). Depending by the location and habitat type, extensive grazing has fewer negative environmental impacts than intensive grazing systems, and if managed appropriately can promote and maintain biodiverse habitats (Zhang et al., 2015).

Grass-based livestock diets

Grass-fed systems are those where livestock is raised either solely or seasonally on pasture with diet supplementation of silage and fodder while overwintering (Clark & Tilman, 2017). Grazed pasture and forage are the single most important feed for ruminants (Wilkinson & Lee, 2018), and feeding more highly digestible grass (herbage or silage) have a number of environmental and human health benefits. In that respect, grass-fed beef was reported to have higher micronutrient concentrations and a fatty acid profile that might lead to improved human health outcomes relative to consumption of grain-fed beef (Daley et al., 2010). Pastures not only provide the lowest unit-cost of production, , but also utilize resources which could not otherwise be used directly for human nutrition, and at the same time return carbon to soil through their manures (Lee et al., 2021).

Grass-based livestock diets can promote: i) soil carbon sequestration (Derner & Schuman, 2007), ii) within-pasture nutrient cycling and reduction of eutrophication (Smith et al., 2013), iii) food security as the system can utilise land not suitable for crop production (Smith et al., 2013), iv) mitigation of methane (CH_4) emission from enteric fermentation (Martin et al., 2010); (Hristov, Oh, et al., 2013), and v) mitigation of manure emissions (Gerber et al., 2013; Hristov, Ott, et al., 2013; Lee et al., 2012).

It is important to note, however, that different ruminants (e.g., cattle, goats, sheep) differ considerably in their level of feed intake, rumen morphology, and physiology (Van Gastelen et al., 2019), which can affect the mitigation possible from grass-based diets. Feeding more highly digestible grass seems to be more effective for dairy and beef cattle than for sheep (Van Gastelen et al., 2019).

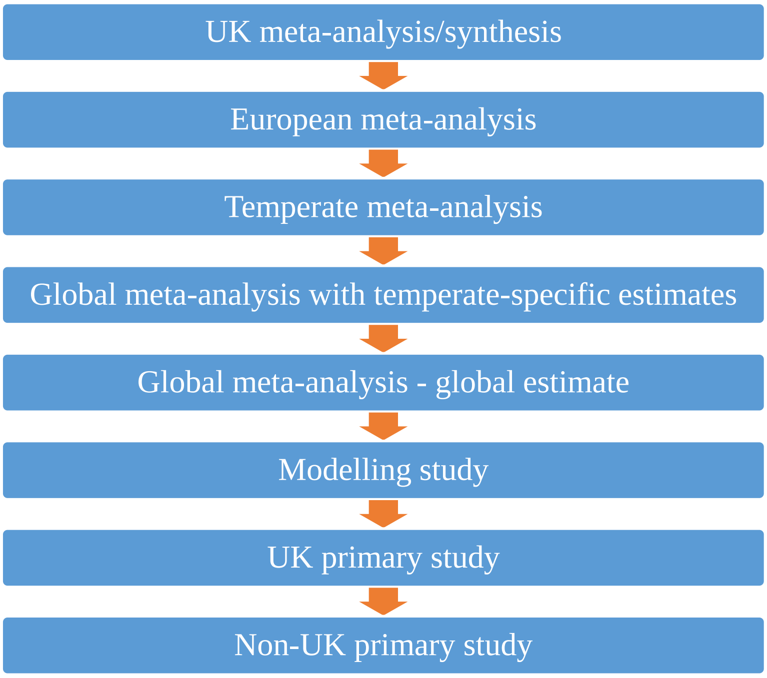
Conversion of temporary leys to permanent pasture

Permanent pasture can be defined as land used to grow grasses or other herbaceous forage that is not cultivated or re-established on an ongoing basis. The conversion from temporary to permanent grassland may involve simple grassland system, or a more diverse species rich system providing favourable outcomes on several ecosystems services (see Section 6 for further detail). Although the composition of permanent grasslands can vary across different regions, their existence depends on ruminant animal production. This means that efforts to reduce CH_4 emissions from ruminant livestock (Gerber et al., 2013) is highly relevant for the future of permanent grasslands.

4 Impact on GHG emissions, above- and below-C stocks, and productivity

In this evidence synthesis, we used Web of Science to identify existing published quantitative meta-analyses and studies on the nineteen AEFPs reported in Table 3. Further details of the search strategy are provided in Appendix A. To apply this data to a UK context, a hierarchy of evidence sources was applied, with parameters extracted from the highest tier at which evidence was available:

Table 3:



Data were extracted from syntheses/studies to spreadsheets (Appendix B), along with relevant meta-data (geographic/climatic generalisability of estimate, number of studies/observations in estimate, study duration, sampling depth (for soil carbon), where in the article the data is located, units of estimates and uncertainty/error parameters, and details of intervention and comparator/control that estimate pertained to). To enable comparison across studies, we harmonised estimates to Response Ratios (RR) (intervention mean/control mean), where a RR of 1 represents no effect of the intervention relative to control, RR < 1 represents a decrease in outcome following adoption of intervention, and RR > 1 represents an increase. A Response Ratio relative to 1 allows the harmonised parameters we present here to be used as a proportional multiplier to baseline emissions, carbon stocks or yield for current conventional land management practices to estimate the impact of adopting a given agroecological practice. Further details of the harmonisation process are provided in Appendix A.

Below we report the potential impact on GHG emissions, soil C stocks, and cropland and livestock production from the implementation of the AEFPs described in Section 3. In Appendix C, we show the extended forest plots outlining the response rations of each AEFP in relation to the studies from where these were extracted. Table 4 summarises the strength of evidence and direction of effect for each AEFP.

Table 4: Strength of evidence and direction of effect for each agroecological farm practice. Note that increased carbon stocks and productivity and reduced GHG emissions are treated as positive. Aboveground biomass not included as interventions to

which this is relevant (buffer strips, agroforestry) inherently increase this. Note it was not possible to assess net GHG balance due to emissions data not always being available across all gases for each intervention.

Intervention		**Comparator**	**GHG emissions**			**Soil carbon stocks**	**Productivity**
			CO_2	CH_4	N_2O		
No-till		Conventional full-inversion tillage	=/+	+	+	=	-/=
Minimum till		Conventional full-inversion tillage	=		=	+	-
Perennial cereal crops		Annual cereal which does not regrow post-harvest				=	-
Pasture cropping		Yields compared to no-till arable with no understory/bare soil. SOC compared to permanent pasture.				+ b	-/=
Liming		No lime addition				=/+	+
Retention of straw		Crop residue removed from field	=		+	+	-/+
Organic manure	FYM	Manufactured or mineral fertiliser	=		=/+	+	-/=
	Slurry		+		+	+	=
	Compost		=		=	+	=
Cover crop		Field left as crop stubble or exposed cultivated soil over winter	=/+		+	=/+	-/=/+
Intercropping		Single crop sown (typically non-legume)					=/+
Legume crops in rotations		Arable rotation does not contain legume crops				=	
Ley-arable		Arable rotation does not include grass-based ley, i.e. continuous arable cropping				+	= d
Silvoarable		No trees present, arable cropping across whole field	=		=	=/+	-/=
Silvopasture		No trees present within or at edge of pasture.				=/+	=/+ f
Vegetated strips	Grass	No margins removed from production, arable cropping across whole field			- a	+	= e
	Hedge				- a	+ c	+ e
	Trees				- a	+	
Rotational grazing		Continuous grazing of pasture by livestock throughout				+	+

	growing season, no rest periods.					
Multispecies leys	Sward entirely composed of grass species			+	+	+
Extensive grazing	Higher stocking density.	+	-	-/+	=/+	**g**
Grass-based diet	Cereal-based supplements (also known as concentrate feeds) included in livestock diet	-	+	=	**i**	**h**
Permanent pasture	Temporary leys, re-established with new seed mix (typically 2-5 years duration)				+	-

Table footnote:

Dark green	strong evidence (1+ temperate estimates) positive effect
Light	weak evidence positive effect
Yellow	mixed evidence no change/ positive effect
Blank	evidence of no change or mixed evidence positive/negative effect
Orange	mixed evidence no change/negative effect
Dark red	strong evidence negative effect
Light red	weak evidence negative effect
Grey	empty cells for no evidence

a. Estimate for buffer strip only relevant to area of buffer, not remaining cropping area
b. Pasture cropping found no change in SOC relative to permanent pasture, so coded here as increase relative to arable
c. SOC increases under buffer strip itself, but also evidence that SOC higher in cropland adjacent to hedge
d. No change in yield in ley-arable systems for years where crop; in non-cropping years when ley growing then complete loss of arable crop yield
e. Yield affect only of crop adjacent to buffer strip; doesn't account for loss yield from reduced cropping area for buffer strip creation
f. Increase if consider tree+forage/livestock growth underneath, no change if consider understory component only
g. No data identified, but if reduce stocking density then obviously decreased production by amount stocking density reduced
h. Although no meta-analyses identified, livestock growth rates are slower on grass-based diet than cereal-based, and grass-based finishing requires more land overall (Clark and Tilman, 2017), so productivity recorded as decreased here.
i. If cropland previously used to grow cereals fed to livestock was converted to grassland and livestock reared on this instead, this would increase soil carbon (cropland to pasture conversion increases SOC, Conant et al. 2017, Guo & Gifford, 2002), although no studies identified which explicitly quantified this scenario.

4.1 Reduced soil disturbance

Reduced tillage intensity

A meta-analysis of long-term experiments in Europe found no effect of no or minimum tillage on carbon dioxide (CO_2) emissions (Sanden et al., 2018), although a global analysis of cool temperate regions identified a significant increase in CO_2 emissions in no-till systems compared to conventional full-inversion tillage (Shakoor et al. 2021) (Fig 2). Nitrous oxide emissions (N_2O) increased under no-till, but did not change in minimum tillage systems, compared to full-inversion tillage (Sanden et al 2018; (Shakoor et al., 2021). CH_4 emissions did not change in no-till systems (Shakoor et al. 2021) (Fig 2).

Soil organic matter did not change under no-till, but increased in minimum tillage systems, compared with full inversion tillage (Sanden et al. 2018; Powlson et al., 2012) (Fig 3). Sanden et al. (2018) found evidence of decreased arable crop yield in no and minimum tillage systems, but another European meta-analysis did not find a negative effect of reduced tillage intensity on yields (Van Den Putte et al., 2010)

(Fig 4), aligning with the results of Sun et al. (2020) for intermediate-humidity environments such as north-west Europe.

Pasture cropping

A field study conducted in New South Wales, Australia, identified no change in SOC stocks under pasture cropping compared to permanent pasture (Badgery et al., 2014) (Fig 3). This represents a substantial improvement over arable cropland, which typically has substantially lower soil carbon than undisturbed permanent pasture. However, arable crops established in permanent pasture have lower yields than crop monocultures (Millar & Badgery, 2009; Lawes et al., 2014), except under low fertiliser regimes where pasture crop yields can match those of no till cropping (Lawes et al., 2014) (Fig 4).

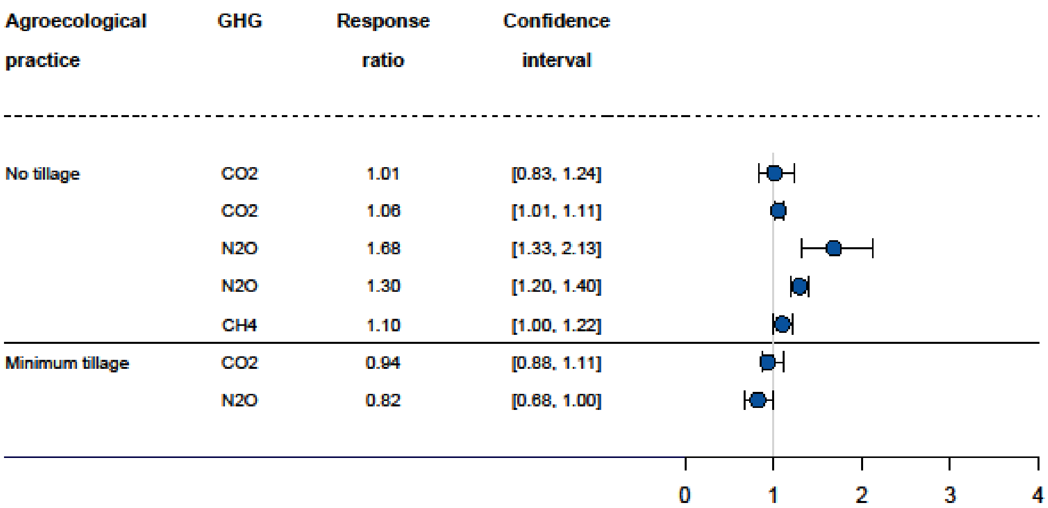
Perennial cereal crops

Perennial cereals are still largely at the development stage, resulting in little data available from field experiments and limited farmer uptake. A field trial of perennial wheat in southern Germany found no change in SOC stocks compared with annual wheat under organic management (Audu et al., 2022) (Fig 3). Hayes et al. (2012) tested a number of perennial wheat cultivars in New South Wales, Australia, and found that while some cultivars re-grew for a second or even third season, harvestable yield greatly declined after the first year.

Liming

A global meta-analysis with estimates specific to temperate regions found that liming increased grassland SOC stocks relative to no lime application, although this increase was only statistically significant when lime was applied in moderate levels (3-5 t ha^{-1}) (Eze et al. 2018) (Fig 2). Liming has also been found to increase grass dry matter yields in areas with moist cool climates (which includes the UK) (Abdalla et al. 2022) (Fig 4). Abdalla et al. (2022) also considered the effects of liming on greenhouse gas emissions, and in the absence of sufficient studies to conduct a meta-analysis, a narrative synthesis of available studies suggested that N_2O and CH_4 emissions either do not change or decrease compared to no lime application, although CO_2 emissions are at risk of increasing.

Figure 2: Impact of no tillage and minimum tillage on greenhouse gas emissions extracted from Sanden et al. (2018), expressed as response ratios (see footnotes).



Footnotes for Figures 2-17:

- Results are presented and plotted as response ratios, calculated as the intervention mean divided by control mean, where a RR of 1 represents no effect of the intervention relative to control (vertical grey line), RR < 1 represents a decrease in outcome following adoption of intervention, and RR > 1 represents an increase. It is important to note that RR is indicative of change and does not indicate that the system will become a sink.Confidence Intervals are 95%. Confidence intervals not presented where error parameters not available in underlying article

- No data was identified for practices not included in a figure
- Dark blue points correspond to temperate meta-analyses (including Europe meta-analyses, temperate meta-analyses, and global analyses with temperate estimates), medium blue points indicated estimates from global meta-analyses without temperate-specific estimates, and light blue points represent modelling estimates (result from modelling simulation or parameter used as model input, typically generated from informal lit review) or primary studies (individual field study, typically non-UK).

Figure 3: Impact of agroecological farming practices that reduce soil disturbance on soil organic carbon, expressed as response ratios (see Fig 2 footnotes). The information on reduced tillage intensity were extracted from Powson et al. (2012), Sanden et al. (2018), and Van Den Putte et al. (2010). Pasture cropping was extracted from Badgerry et al. (2014). Perennial cereals was extracted using Audu et al. (2022). Liming was extracted from Eze et al. (2018).

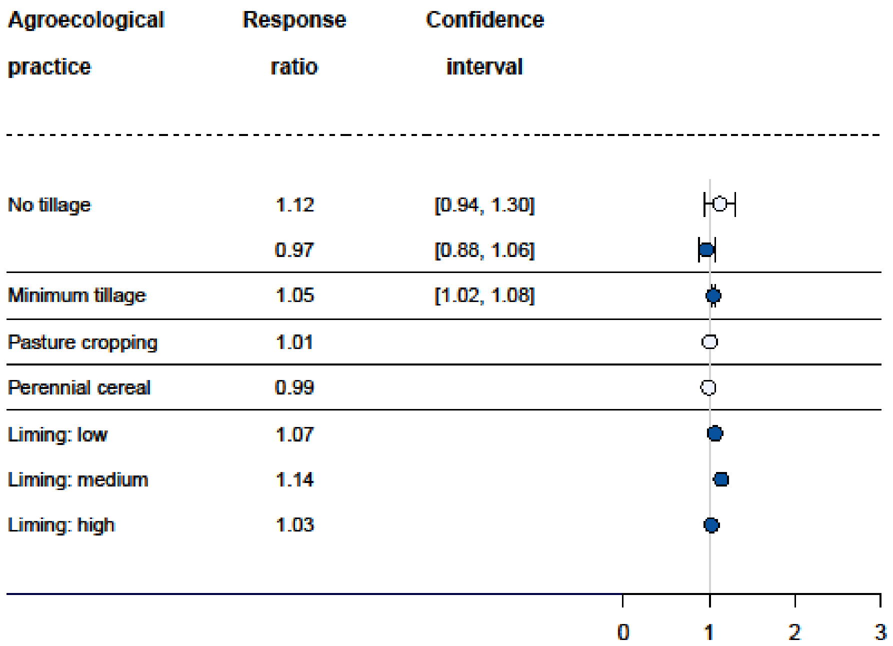
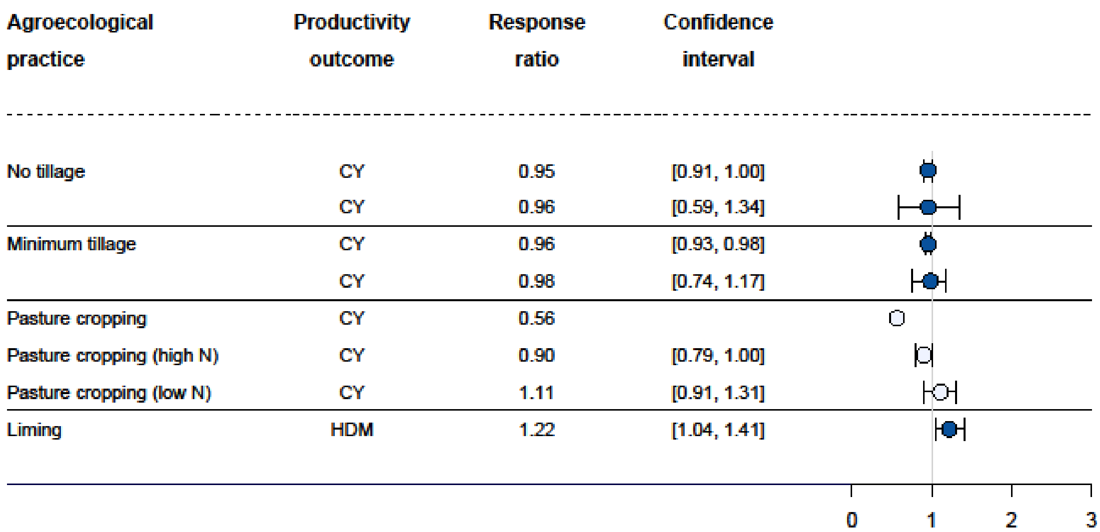


Figure 4: Impact of agroecological farming practices that reduce soil disturbance on agricultural productivity, expressed as response ratios (see Fig 2 footnotes). The information on reduced tillage intensity were extracted from Sanden et al. (2018), and Van Den Putte et al. (2010). Pasture cropping was extracted from Lawes et al. (2014). Perennial cereals was extracted using Hayes et al. (2012). Liming was extracted from Abdalla et al. (2022).



4.2 Organic inputs

Organic manure

Farmyard manure (FYM) had no effect on CO_2 or N_2O emissions compared with inorganic fertilisers with a similar total N content in European long-term experiments (Sanden et al. 2018), although a global meta-analysis found FYM increased N_2O in warm temperate regions which included the UK (Zhou et al., 2017) (Fig 5). FYM significantly increased SOC in both the UK (compared to no FYM) and Europe (compared to equivalent N fertilisation) (Powlson et al. 2012; Sanden et al. 2018) (Fig 6). Crop yields were lower under FYM applications in Europe (Sanden et al. 2018) but did not differ globally (Shang et al., 2021) (Fig 7), in both instances compared to inorganic fertiliser with similar N content.

Slurry applications in Europe resulted in higher emissions of both CO_2 and N_2O compared to equivalent N fertilisation (Sanden et al. 2018) (Fig 5). However, slurry did increase SOC content (Fig 6) and did not impact arable crop yields (Sanden et al. 2018) (Fig 7).

Use of compost instead of inorganic fertilisers did not impact CO_2 or N_2O emissions (Fig 5) and increased SOC content in a European context (Sanden et al. 2018) (Fig 5). European and global meta-analyses found no effect of compost vs inorganic fertiliser on crop yields (Sanden et al. 2018, Shang et al. 2021) (Fig 7).

Retention of straw

Crop residue retention did not affect CO_2 emissions, but significantly increased N_2O emissions compared to residue removal in both European long-term experiments and global meta-analysis with temperate estimates (Sanden et al. 2018; Li et al., 2021) (Fig 5), although this does not account for removed residues potentially being returned later as part of FYM. Straw retention increased SOC in both the UK and Europe, compared to straw removal (Powlson et al. 2012, Sanden et al. 2018) (Fig 6). The effect of straw retention on arable crop yield are mixed; lower yields were identified in Europe compared to straw removal (Sanden et al. 2018) but globally yields increased when residues were retained (Liu et al., 2014) (Fig 7).

Figure 5: Impact of agroecological farming practices which increase organic inputs on greenhouse gas emissions, expressed as response ratios (see Fig 2 footnotes). The information on retention of straw was extracted from Sanden et al. (2018), Li et al., (2021). Organic manure was obtained using Powlson et al. (2012), Sanden et al. (2018), and Shang et al., (2021).

Agroecological practice	GHG	Response ratio	Confidence interval
Retention of straw	CO2	1.12	[0.94, 1.33]
	N2O	1.63	[1.18, 2.25]
	N2O	1.43	[1.25, 1.68]
Organic manure: FYM	CO2	1.04	[0.92, 1.18]
	N2O	0.62	[0.25, 1.54]
	N2O	1.34	[1.07, 1.65]
Organic manure: slurry	CO2	1.31	[1.16, 1.48]
	N2O	2.98	[1.02, 8.72]
Organic manure: compost	CO2	1.24	[0.80, 1.92]
	N2O	2.35	[0.40, 13.79]

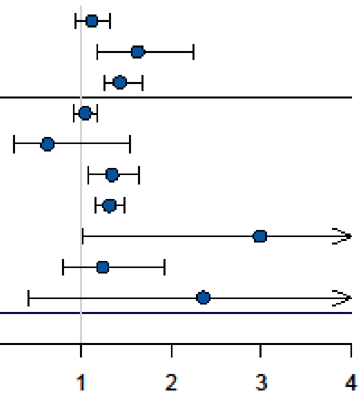


Figure 6: Impact of agroecological farming practices which increase organic inputs on soil organic carbon extracted from Powlson et al. (2012) and Sanden et al. (2018), expressed as response ratios (see Fig 2 footnotes).

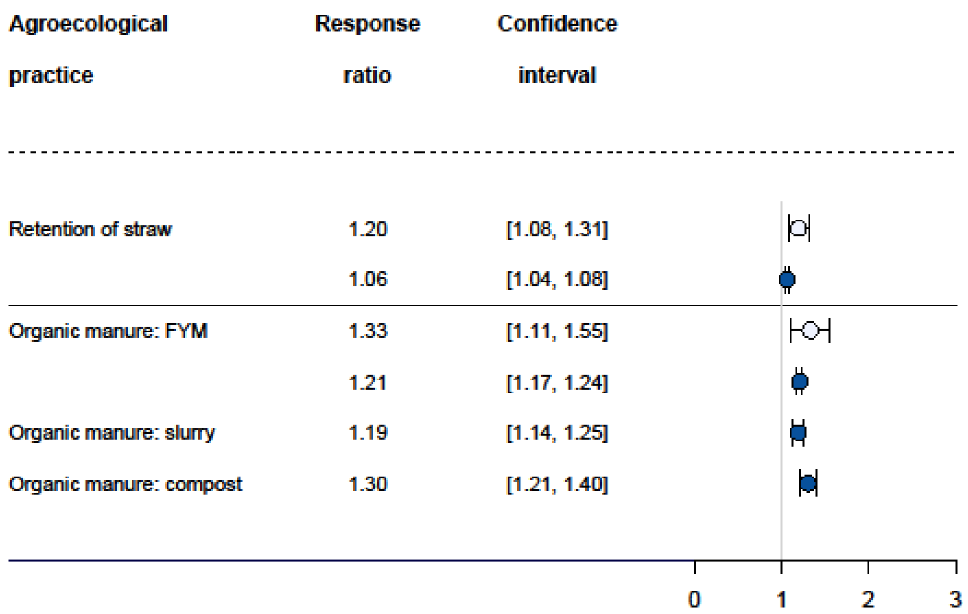
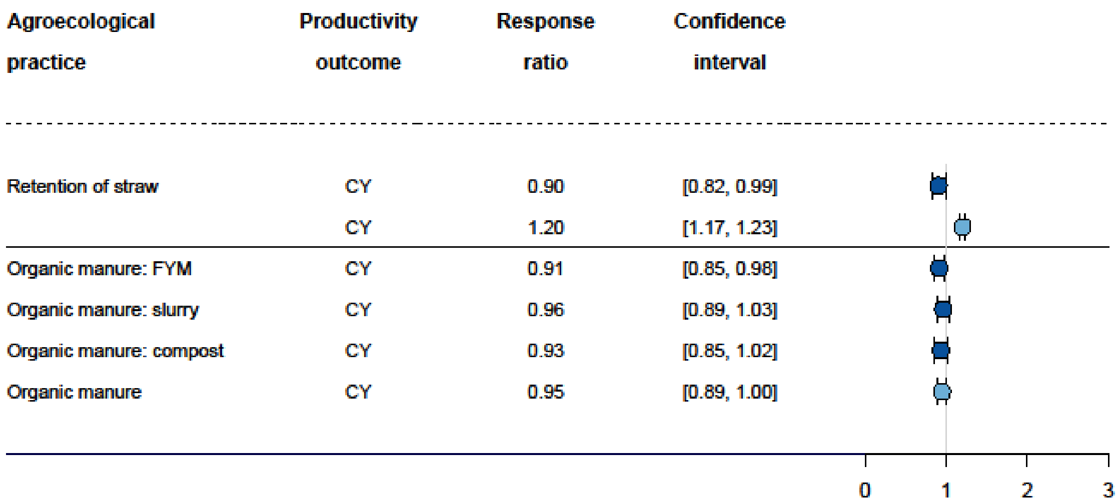


Figure 7: Impact of agroecological farming practices which increase organic inputs on agricultural productivity, expressed as response ratios (see Fig 2 footnotes). Data were extracted from Powlson et al. (2012), Sanden et al. (2018), and Shang et al. (2017).



4.3 Diverse crop rotations

Cover crops

Cover cropping did not affect CO_2 emissions in European long-term experiments, but significantly increased N_2O emissions (Sanden et al. 2018) (Fig 8). Evidence from European meta-analysis indicates that cover crops do not significantly change SOC compared to crop rotations without cover crops (i.e. crop stubble left fallow over winter) (Sanden et al. 2018), according with results from some global studies e.g. (Abdalla et al., 2019) (Fig 9) although other temperate and global analyses find a positive effect of cover cropping on soil carbon where the growing window is sufficiently long (McClelland 2021; Poeplau 2015). Cover crops (legume, non-legume and mixed) also did not affect arable crop yields in European or global meta-analyses (Sanden et al., 2018; Abdalla et al., 2019), although faster growing catch crops increased subsequent crop yields (Sanden et al., 2018) (Fig 10).

Intercropping

A meta-analysis of European studies found that both substitutive (sum of relative sowing densities equivalent to sole crop) and additive (sum of relative sowing densities greater than sole crop) intercropping resulted in overall higher crop yields than monoculture cropping (Mahmoud et al., 2022) (Fig 10). Furthermore, undersowing arable crops with a non-legume or mixed catch crop increased yields compared to monoculture, but undersowing with a legume catch crop decreased yields, according to a meta-analysis of Nordic studies (Valkama et al., 2015) (Fig 10).

Legume crops included in rotation

Meta-analyses focused on the Americas found mixed effects of including legume crops in arable rotations on SOC; McDaniel et al. (2014) identified an increase in SOC concentration when one or two legumes were including in a rotation, whereas West & Post (2002) found no effect on SOC sequestration rate under a maize-soybean rotation (soybean is leguminous) compared to continuous maize (Fig 9).

Ley-arable

Temperate and global meta-analyses have identified increased SOC concentration and sequestration rate when grass-based leys are incorporated into arable rotations (Jordon et al., 2021; Conant et al., 2017; West et al., 2002) (Fig 9). Jordon et al. (2021) found no effect of ley presence on the yield of subsequent arable crops (Fig 10). The absence of crops during the ley phase results in complete arable yield loss in these years, although the ley is typically grazed by livestock or cut for forage or feedstock representing an alternative form of production.

Figure 8: Impact of cover crop on greenhouse gas emissions extracted from Sanden et al. (2018), expressed as response ratios (see Fig 2 footnotes).

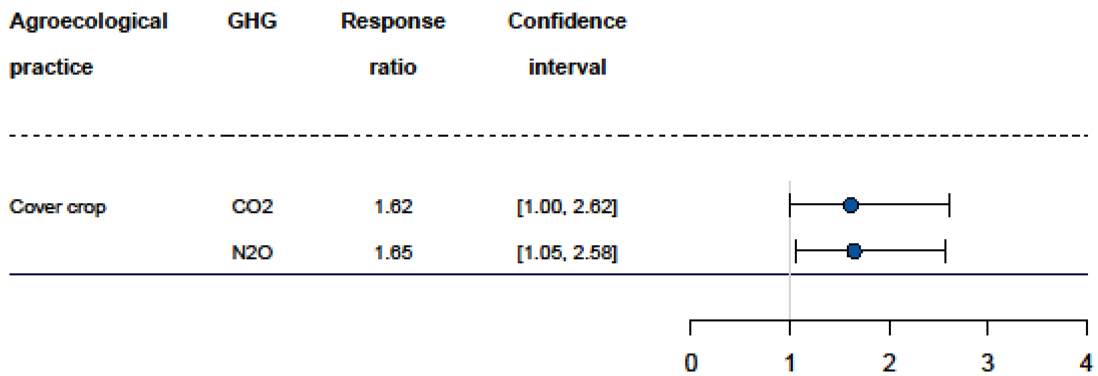


Figure 9: Impact of agroecological farming practices which diversify crop rotations on soil organic carbon, expressed as response ratios (see Fig 2 footnotes). Cover crop was extracted from Sanden et al. (2018) and Abdalla et al., (2019). Legume crops from Mcdaniel et al. (2014) and West & Post (2002). Ley-arable was extracted from Jordon et al. (2021); Conant et al. (2017) and West et al. (2002).

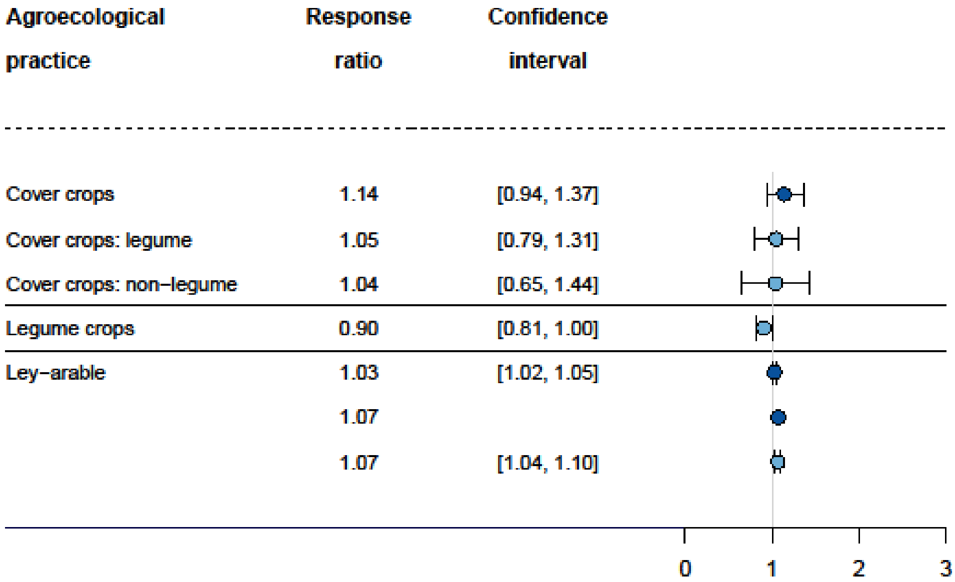
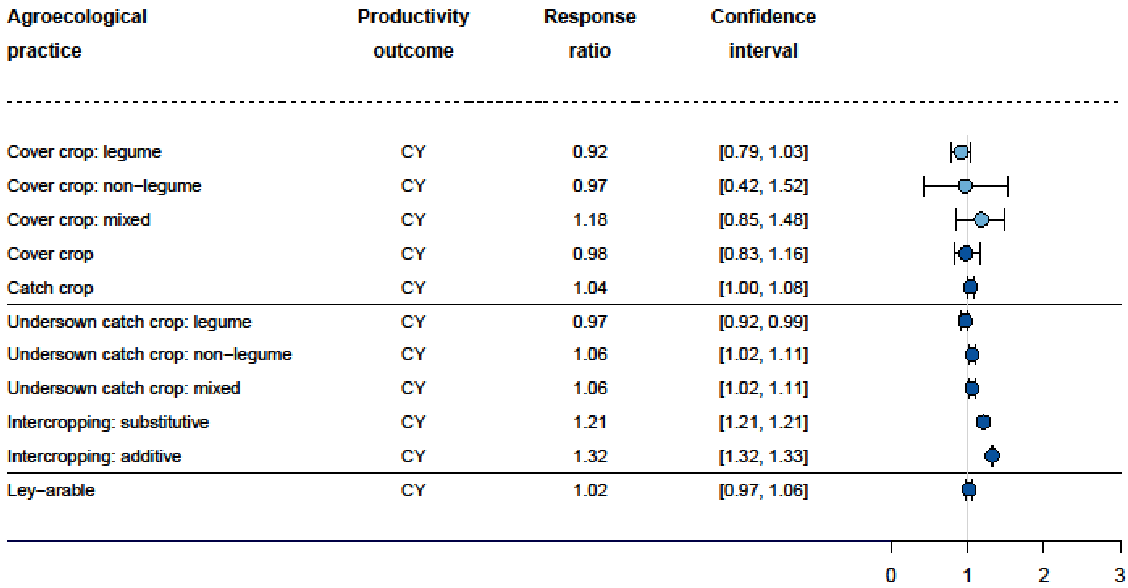


Figure 10: Impact of agroecological farming practices which diversify crop rotations on agricultural productivity, expressed as response ratios (see Fig 2 footnotes). Cover crop was extracted from Sanden et al. (2018) and Abdalla et al., (2019). Ley-arable was extracted from Jordon et al. (2021). Intercropping was extracted using Mahmoud et al. (2022), and Valkama et al. (2015).



4.4 Multifunctional land-use systems

Silvoarable and Silvopasture (Agroforestry)

Smith et al. (2008) estimated that silvoarable systems would have no impact on CO_2 or N_2O emissions compared to treeless cropping systems (Fig 11). The impact of temperate agroforestry systems on soil carbon is mixed; Ivezic et al. (2022) found that silvoarable systems do not affect SOC compared to cropland without trees and Chatterjee et al. (2018) found that SOC in silvopasture systems does not differ to pasture without trees, whereas Mayer et al., (2022) found that both silvoarable and silvopasture have higher SOC stocks than their respective comparators (Fig 12). Global analyses of cropland and pasture agroforestry found both SOC and aboveground carbon stocks significantly increased compared to cropland/pasture without trees (Ma et al., 2020; Kim et al., 2016) (Fig 12, 13).

A European meta-analysis of alley cropping silvoarable systems found a significant reduction in crop yields per area cropped compared to treeless cropping systems due to the negative effect of shading (Ivezic et al. 2021), although a separate European meta-analysis across all types of agroforestry found no effect on food production (Torralba et al., 2016) (Fig 14). Temperate silvopasture systems have been found to overall increase productivity compared to separate pasture and forestry systems (Pent, 2020) (Fig 14), although this productivity increase includes the tree component (i.e. diversification) rather than representing an absolute increase in pasture or animal production.

Vegetated strips

Falloon et al. (2006) estimated, in a UK-based literature review, that field-edge vegetated strips containing grass, hedgerow or trees would all have lower N_2O emissions than the surrounding cropland (Fig 11). UK estimates and temperate meta-analyses also indicate that grass, hedgerow and tree vegetated strips have higher SOC and aboveground carbon stocks than surrounding cropland (Falloon et al. 2004; Van Vooren et al., 2018; Drexler et al., 2021; Mayer et al., 2022) (Fig 12, 13). In addition, Van Vooren et al. (2017) find that hedgerows increase SOC stocks of adjacent cropland up to a D/H ratio (distance from hedge divided by height of hedge) of 4.3 away from the hedge. (Fig 14).

Figure 11: Impact of vegetated strips and silvoarable systems on greenhouse gas emissions, expressed as response ratios (see Fig 2 footnotes). The information on silvoarable was extracted from Smith et al. (2008). Vegetated strips were extracted from Falloon et al. (2006).

Agroecological practice	GHG	Response ratio	Confidence interval
Silvoarable	CH4	1.00	[1.00, 1.00]
	N2O	1.03	[0.94, 1.14]
Vegetated strips: grass	N2O	0.08	
Vegetated strips: hedge	N2O	0.10	
Vegetated strips: trees	N2O	0.13	

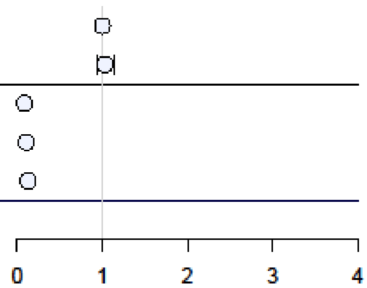


Figure 12: Impact of create multifunctional land-use systems on soil organic carbon, expressed as response ratios (see Fig 2 footnotes). Silvoarable and silvopasture systems were extracted from Ivezic et al. (2022), Mayer et al., (2022), Ma et al. (2020),

Kim et al. (2016). Vegetated strips were extracted from Falloon et al. (2004), Van Vooren et al. (2018) Drexler et al. (2021) and Mayer et al. (2022).

Agroecological practice	Response ratio	Confidence interval
Silvoarable	1.03	[0.61, 1.40]
	1.01	[0.07, 1.60]
	0.90	[0.61, 1.08]
Agroforestry	1.33	[1.18, 1.47]
Silvopasture	1.41	[0.59, 2.11]
	1.14	[0.90, 1.37]
	1.02	[0.89, 1.15]
	0.95	[0.83, 1.07]
Vegetated strips: grass	1.37	
	1.29	
Vegetated strips: hedge	1.32	[1.15, 1.51]
	1.24	[0.25, 3.10]
	1.45	[1.05, 1.84]
	1.28	
Arable 1.5m from hedge	1.14	
Arable 3m from hedge	1.09	
Vegetated strips: trees	1.26	

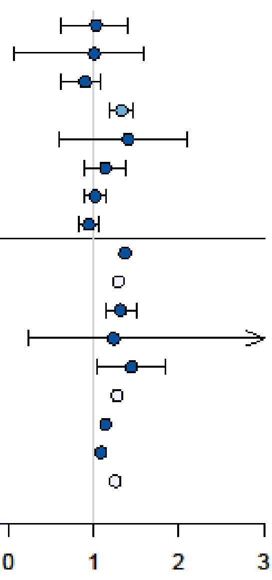


Figure 13: Impact of agroecological farming practices which create multifunctional land-use systems on aboveground carbon sequestration, expressed as annual carbon sequestration rate. Agroforestry and silvopasture systems were extracted from Ma et al. (2020), Kim et al., (2016). Vegetated strips were extracted from Falloon et al. (2004) and Drexler et al. (2021).

Agroecological practice	Sequestration (t C/ha/yr)	Confidence interval (95%)
Agroforestry	3.08	[1.71, 4.40]
Silvopasture	4.60	[2.20, 7.00]
Vegetated strips: grass	0.20	
Vegetated strips: hedge	2.35	[2.00, 2.70]
	1.00	
Vegetated strips: trees	2.80	

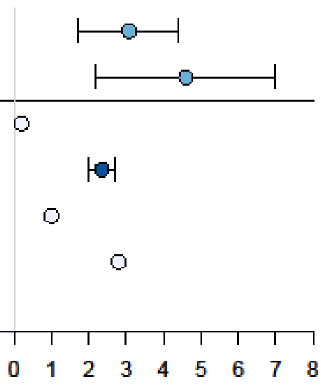
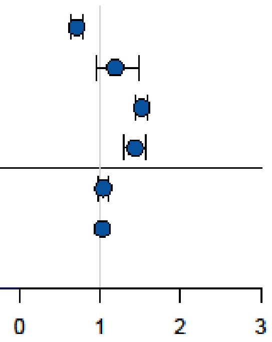


Figure 14: Impact of multifunctional land-use systems on agricultural productivity, expressed as response ratios (see Fig 2 footnotes). Agroforestry and silvopasture systems were extracted from Ivezic et al. (2021), Torralba et al., (2016), Pent. (2020). Vegetated strips were extracted from Van Vooren et al. (2017) and Lowe et al. (2021).

Agroecological practice	Productivity outcome	Response ratio	Confidence interval
Silvoarable	CY	0.71	[0.64, 0.78]
Agroforestry		1.19	[0.95, 1.48]
Silvopasture	LER	1.52	[1.44, 1.60]
	LER	1.44	[1.30, 1.58]
Vegetated strips: pollinator mix	CY	1.04	[0.98, 1.10]
Arable 30m from hedge	CY	1.03	



4.5 Pasture productivity

Rotational grazing

A global meta-analysis identified a significant increase in SOC stocks under rotational grazing compared to continuous grazing systems (Byrnes et al., 2018), but there is no effect on aboveground carbon stocks in herbage biomass (Mcdonald et al., 2019) (Fig 16). A meta-analysis of studies from temperate oceanic regions found higher livestock growth rates (daily liveweight gain) under rotational grazing compared to continuous grazing with no rest periods (Jordon et al., under review) but no effect was found in a global analysis (McDonald et al. 2019) (Fig 17). Herbage dry matter yields (Jordon et al. under review) and animal yields per hectare (McDonald et al., 2019) also increase under rotational grazing systems compared to continuous grazing (Fig 17).

Multispecies leys

A global meta-analysis found that including legumes in pasture swards increased SOC stocks, compared to grass only swards (Conant et al., 2017) (Fig 16). A modelling study also found that legume inclusion at 20% of the sward would increase soil carbon sequestration without affecting N_2O emissions in Western Europe (Henderson et al., 2015) (Fig 15). Multispecies swards that include perennial forbs in addition to grass and clover species increase herbage dry matter yields, livestock growth rates and dairy milk yields in temperate regions (Jordon et al. under review; McCarthy et al. 2020) (Fig 17).

Figure 15: Impact of agroecological farming practices which increase pasture productivity or extensify livestock yields on greenhouse gas emissions, expressed as response ratios (see Fig 2 footnotes). The information on extensive livestock systems was obtained from Tang et al. (2019), and Zhou et al. (2017). Grass-based diets was extracted from Lynch et al. (2019), Smith et al. (2008) and Clark & Tilman (2017). Permanente pasture was extracted from Smith et al. (2020).

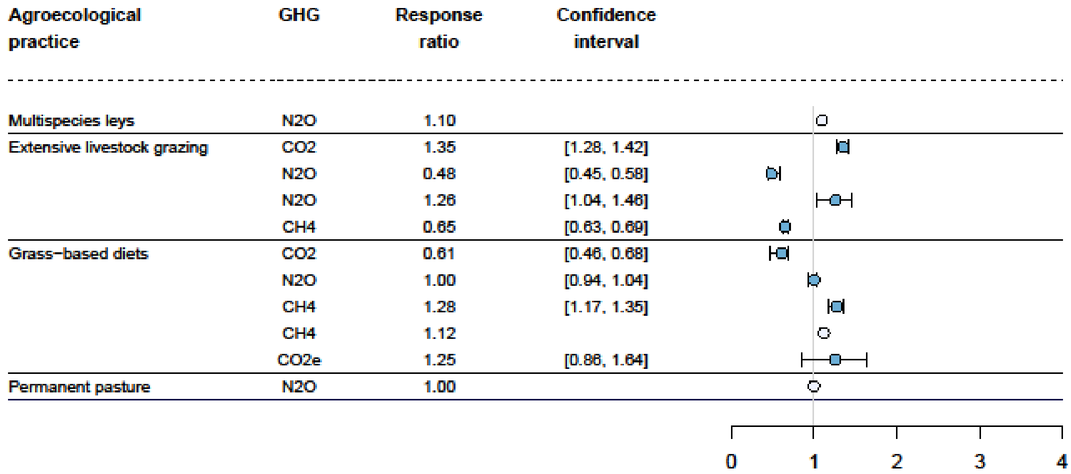


Figure 16: Impact of agroecological farming practices which increase pasture productivity or extensify livestock yields on soil organic carbon, expressed as response ratios (see Fig 2 footnotes). Rotational grazing was extracted using Byrnes et al. (2018), and McDonald et al. (2019). Multispecies leys was extracted from Conant et al. (2017) and Henderson et al. (2015). The inforamtion on extensive livestock grazing was obtained from Tang et al. (2019), Zhan et al. (2020), Eze et al., (2018) and Byrnes et al. (2018). Permanent pasture was extracted from Smith et al. (2010).

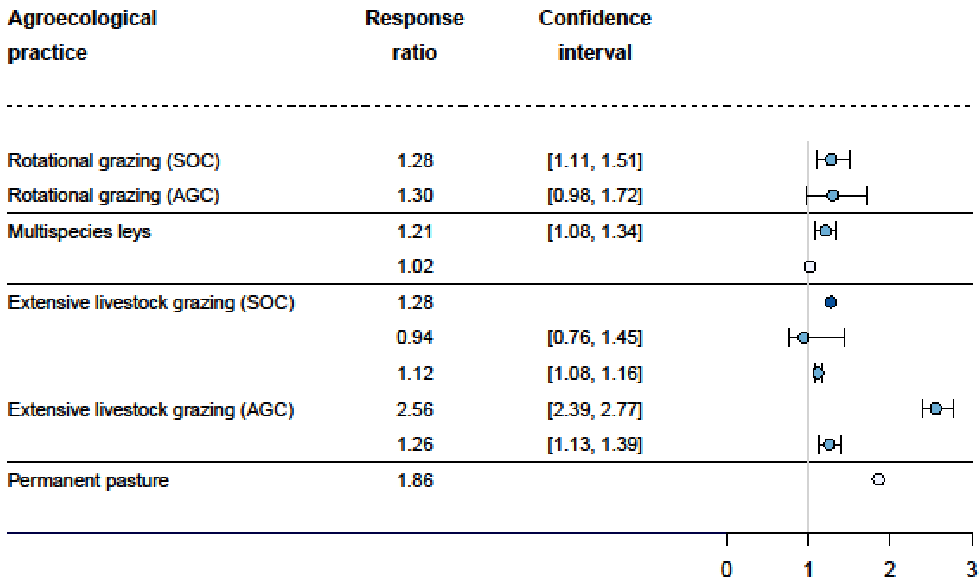
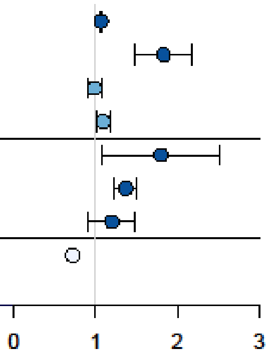


Figure 17: Impact of agroecological farming practices which increase pasture productivity or extensify livestock yields on agricultural productivity, expressed as response ratios (see Fig 2 footnotes). The information on rotational grazing and

Multispecies leys was extracted using Jordon et al. (2022), McCarthy et al. (2019) and McDonald et al. (2019). Permanent pasture was extracted from Qi et al. (2018).

Agroecological practice	Productivity outcome	Response ratio	Confidence interval
Rotational grazing	HDM	1.07	[1.06, 1.07]
	LWG	1.83	[1.49, 2.18]
	LWG	0.99	[0.92, 1.07]
		1.09	[1.01, 1.19]
Multispecies ley	HDM	1.80	[1.08, 2.52]
	LWG	1.37	[1.23, 1.51]
	MY	1.20	[0.90, 1.49]
Permanent pasture	HDM	0.72	



4.6 Extensive livestock systems

Extensive grazing (reduced stocking density)

A meta-analysis focused on studies in China found that reduced stocking density significantly increases soil CO_2 emissions and decreases soil CH_4 emissions compared to high stocking densities (Tang et al., 2019) (Fig 15). There is conflicting evidence on N_2O emissions; Tang et al. identify a significant increase, whereas a separate global meta-analysis finds decreased N_2O emissions when stocking density is reduced (Zhou et al., 2017) (Fig 15).

Abdalla et al., (2018) found that although livestock grazing reduces SOC compared to no grazing in temperate regions, there is no difference between light vs heavy grazing in this effect (Fig 16). However, other global meta-analyses find that SOC is significantly higher under light stocking densities than heavy grazing (Eze et al., 2018; Byrnes et al., 2018) (Fig 16). Aboveground carbon stocks in herbage biomass are also increased when stocking density is reduced (Tang et al., 2019; Zhan et al., 2020) (Fig 16).

Grass-based livestock diets

A systematic review of lifecycle assessments found that grass-fed beef has significantly lower CO_2 emissions than beef from non-grass-fed systems (i.e. component of diet comprised of cereal grains or other concentrate feeds) (Lynch, 2019) (Fig 15). CH_4 emissions are likely increased in grass-fed systems (Lynch 2019, Smith et al. 2008), but N_2O emissions (Lynch, 2019) and overall greenhouse gas emissions (CO_2 equivalent) (Clark & Tilman, 2017) are no different between grass-fed and grain-fed beef systems (Fig 15).

Conversion of temporary leys to permanent pasture

(Smith et al., 2010) estimated that conversion from temporary leys to permanent pasture in Great Britain would have no effect on N_2O emissions (Fig 15) and would increase SOC stocks (Fig 16), based on results from Rothamsted Research field trials. Forage dry matter yield is projected to decrease if UK temporary leys were converted to permanent pastures (Qi et al., 2018) (Fig 17).

5 Co-benefits and trade-offs

5.1 Summary of co-benefits and trade-offs

In this section, we qualitatively summarise the results of the literature review on potential positive (co-benefits) and negative (trade-offs) effects that each AEFP can generate on 27 indicators spread across six agroecological principles (Table 4). These results, together with the three main indicators assessed in Section 4 constitute the overall impact of the AEFPs here reviewed.

Agroecological systems such as agroforestry or intercropping still lack scientific evidence (Struik & Kuyper, 2017). For example, it is still difficult to directly link biodiversity to agricultural intensification, resource use efficiency or resilience (Bretagnolle et al., 2011; Inchausti & Bretagnolle, 2005). In addition, it is uncertain whether biodiversity loss results more from the cropping intensification at field level or from the uniformity of the landscapes (Lemaire et al., 2015). However, a potential agroecological transition requires multiple objectives to be considered and be aware of the fact that the implementation of different AEFPs can have multiple outcomes on various indicators at farm and landscape level. Although the implementation of some AEFPs can be compatible with others (Table 1), in general their benefits and trade-offs are not additive when the effect is related to the same indicators (Smith et al., 2008). It is plausible to assume that in the implementation of multiple AEFPs the outcomes on different indicators are cumulative (see examples described in the legend of Table 5), and when the generated benefit targets the same indicator the overall confidence of the outcome is higher. In this context, in Table 5 we provide an indication of the confidence related to their outcomes, which is based on the evidence found in the literature.

Table 5: Summary of the co-benefits and trade-offs generated from the application of agroecological farm practices on sustainable indicators that correspond to specific principle of agroecology. Positive or negative outcomes correspond to the increase or decrease effect on indicators. For each agroecological practice, a panel comprising the report authors scored each outcome at the farm level based on the expected change and knowledge derived from this literature review. Using a six-point scale (ranging from a positive effect with high confidence to a negative effect with high confidence, see legend), each score shows the expected impact if a conventional farming system were to implement a particular agroecological practice. For example, Rotational grazing has a positive effect on reducing land-use requirement, or Organic manure has a negative effect on Nutrient management as it increase the difficulty in optimising nutrient availability in soils based on plant demands. In some instances, the effects from the practices can be modified by the simultaneous use of another practice. For instance, reduced tillage can negatively affect bulk density (i.e. increase BD), but the simultaneous use of Crop residue retention can change the effect of Reduce tillage to positive. Liming does not represent an agroecological practice in itself. Depending by the substitution of practice implemented, liming can support the agroecological transition remediating the negative impact caused by conventional agriculture.

		Agroecological principles and their indicators																								
Agroecological practices		Input reduction			Soil health							Animal health			Biodiversity			Recycling	Synergies							
		Pesticide use	Herbicide use	Land-use requirement	Water holding	Soil cover/protection	Soil structure	Bulk density	Soil compaction	Soil erosion/run-off	Soil acidification	Macronutrient in diet	Digestibility diet	Livestock wellfare	Species richness	Earthworm abundance	Arthropods abundance	Nutrient management	Water quality	Air quality	Nutrient losses	Weed control	Pathogen control	Resilience	Fire risk	Wind protection
Reduced soil disturbance	No till (NT)	N-M	N-M		P-H		P-M	N-M	N-M	N-M						P-H			N-L		N-M	N-M		P-L		
	Minimum till (MT)	N-L	N-L		P-H		P-M	N-L / P-M (RS)	N-L / P-M (RS)	N-L / P-M (RS)						P-M					N-L / P-M (RS)	N-L		P-L		
	Perennial cereals (PeC)	N-M			P-M	P-H	P-H			P-H		P-H			P-M			P-L	P-H	P-M	P-M	N-L		P-L		
	Pasture cropping (PaC)				P-H	P-H	P-H			P-H					P-L			P-M				P-H				
	Liming (L)						P-M				P-H					P-M		P-H								
Organic inputs	Retention of Straw (RS)				P-L	P-H	P-H	P-H	P-L	P-H					P-H	P-H	P-M	N-L			P-M					
	Organic manure (OM)						P-L	P-H		P-L	N-L					P-H		N-H	N-M	N-H			N-L			
Diverse crop rotations	Cover crop (CC)	P-L (SR)	P-L (SR)		P-H	P-H		P-L / P-M (MT)	P-L / P-M (MT)	P-L / P-M (MT)		P-H			P-M	P-M		P-H	P-L	P-L	P-L	P-L (SR)		P-M		
	Intercropping (I)	P-H	P-H		P-H	P-H				P-H					P-M			P-H			P-M	P-M	P-L	P-L		
	Legume crops (LC)	P-L (SR)	P-L (SR)		P-H			P-L	P-L	P-L		P-H			P-M			P-H			N-L	P-L (SR)		P-M		
	Ley-arable (LA)		P-M				P-M			P-M / N (NT)					P-M	P-M		P-M	P-L	P-L	P-L	P-H		P-L		
Multifuncional land-use	Silvoarable (SP)	N-L													P-H	P-H	P-H						N-M	P-H		P-H
	Silvopasture (SA)								N-M			P-H		P-H	P-H		P-H		P-H	P-H				P-H		P-H
	Vegetated strips (VS)	P-L	P-L	N-L	P-H	P-H				P-H					P-H	P-H	P-H	P-M	P-H	P-H	P-H	N-L	N-L	P-L		P-L
Pasture productivity	Rotational grazing (RG)			P-H	P-L (CR)	P-L		P-L (CR)	N-H	N-H			P-L		P-H			P-M	N-L	N-L	N-H	P-M				
	Multispecies leys (ML)		P-M		P-H	P-H						P-H		P-H	P-H	P-H		P-M	P-L		P-H			P-H		
Extensive livestock	Extensive grazing (EG)			N-H		P-H			P-M	P-M			N-L	P-H	P-L						P-M			P-L	P-L	
	Grass-based livestock diets (GD)			N-H								N-H	N-H	P-H												
	From temporary to permanent pasture (PP)			N-L	P-L	P-M				P-M			N-L		P-M	P-H		P-L	P-L	P-L				P-M		

P-H	Positive effect with high confidence	SR = Short rotation
P-M	Positive effect with medium confidence	T= Tillage
P-L	Positive effect with low confidence	CR = Continuous rotation
N-L	Negative effect with low confidence	
N-M	Negative effect with medium confidence	
N-H	Negative effect with high confidence	
	Unknown or not available in literature	
	Not applicable	

Footnote:
The indicator resilience corresponds to the capacity to resist and recover from perturbation and maintain the flow of ecosystem services. The concept of 'resilience' is context specific and difficult to quantify. Although other indicators are essential for assessing the long-term sustainability of the practices, in the category "synergies" we summarise the overall impact of different indicators as a functional resilience to stressors, such as climate change, invasive species, etc.
In Rotational grazing the positive effect on bulk density and negative effect on soil compaction is determined by management: if rotational grazing is managed correctly it shouldn't negatively affect compaction or soil erosion/run-off.
Cover crops may increase herbicide use if they are chemically terminated.
Soil compaction and bulk density are related to each other. However, in Table 5 these two indicators are separated to highlight the direct effect of the agricultural practice.
Ley arable reduces soil erosion/run-off, as the grass-based ley holds the soil together over winter unlike an arable fallow/stubble.
Ley arable might have a negative effect on soil erosion/run-off if combined with no-till.

5.2 Reduced soil disturbance

Reduced tillage intensity

Co-benefits:

Briones and Schmidt (2017) showed that in general reducing tillage intensity promotes earthworm abundance and biomass. A recent meta-analysis of Sanden et al. (2018) showed that minimum tillage significantly enhanced earthworm number and biomass by 33% and 68%, respectively, due to reduced soil disturbance preserving earthworms and their burrows. However, the impact of reduced tillage might depend on the agricultural management conditions (Chan, 2001). In organic farming, for instance, the absence of agrochemicals, diversified crop rotations and the use of organic fertilizers may promote earthworms irrespective of the tillage methods used (Bertrand et al., 2015).

Sanden et al. (2018) indicated that no-till significantly increased available P in soil by 25%, as well as earthworm number and biomass by 84% and 370% respectively. Non-inversion tillage significantly increased nutrient contents such as N and Potassium by 7% and 43% respectively. In addition, non-inversion tillage significantly enhanced earthworm number and biomass by 33% and 68%, respectively.

There is evidence that minimum tillage can be used to rectify soil compaction (Langmaack et al., 2002; Langmaack et al., 1999) especially if used in conjunction with subsoiling and cover cropping (Raper et al., 2000). Sanden et al. (2018) did not find any significant effect on soil bulk density, but highlighted a significant decrease in run-off and sediment yield by 28% and 63%, respectively.

Trade-offs:

Long-term use of minimum tillage, and in particular no-till, can lead to soil compaction and thereby lower yields, increased runoff and poor infiltration (Hussain et al., 1998); (Ferreras et al., 2000); (Raper et al., 2000). There are opposing mechanisms underlying the effect of minimum tillage on runoff. On the one hand, minimum tillage with residue retention can increase soil surface roughness, prevent surface crusting and sealing, and improve pore continuity, thus increasing infiltration and reducing runoff (Blanco-Canqui & Lal, 2009); (Armand et al., 2009). On the other hand, continuous minimum tillage may increase bulk density and decrease macroporosity, thereby decreasing sorptivity and hydraulic conductivity (Alvarez and Steinbach (Alvarez & Steinbach, 2009); (Fasinmirin & Reichert, 2011).

If compaction occurs as a consequence of long-term use of minimum tillage, P can accumulate on the soil surface increasing loss via runoff (J. H. Moos, 2017). For instance, by increasing the use of minimum tillage over a period of 20-year from 5 to 50% of the planted area, soil loss was reduced by 49% along with the transport of P (Richards and Baker,1998), but the concentration in runoff was higher, leading to an overall loss that was 1.7–2.7 times greater (Gaynor & Findlay, 1995).

Higher nitrate (NO_3) leaching losses and deeper nitrate infiltration occurred with no-till (Eck & Jones, 1992). Similarly, Kandeler and Bohm (1996) reported higher N-mineralisation under no-till and minimum tillage. With no-till NO_3–N concentrations were lower, but under no-till the total NO_3–N losses were higher because the total volume of water moving through the soil was higher compared to conventional tillage (Kanwar, 1997).

Minimum tillage can negatively influence the environmental impact of pesticides in two ways. Firstly, through modification of the soil structure and functional processes that consequently affect the fate of pesticides once applied. Secondly, by influencing the levels of crop pests, diseases and weeds and thereby the need for pesticides. Minimum tillage may increase the risk of leaching, particularly of herbicides because usage may increase when combating grass weeds, especially during the early transition years, but may eventually be lower (J. H. Moos, 2017). Moreover, the increase in soil macropores facilitates more rapid movement of water and the pesticides into water courses (Harris et al., 1993); (Kamau et al., 1996; Kanwar, 1997).

Perennial cereal crops

Co-benefits:

While cereal grain cropping systems target mainly high productivity, perennial grain crop systems show some potential to add environmental benefits to the agricultural system (Duchene et al., 2019). The perennial nature of these systems means that there is surface cover during the entire year and a more persistent root system. Both of which bring a strong reduction in soil erosion (Ryan et al., 2018). Additionally, the well-developed root system and reduced disturbances improve the soil health by enhancing the porosity, microbial activity, and aggregate stability (Crews et al., 2016; Crews and Brookes, 2014; Culman 2010; Rasche et al., 2017). Also, nutrient holding capacity is increased in perennial grain crop systems, compared to annual crops. This shows potential for improving nutrient use efficiency and reduce N leaching (Jungers et al. 2019). Consequently, this improves the water quality (Culman et al., 2013).

Perennial grain crops have a different nutritional profile (Duchene et al., 2019). Tyl and Ismael (2019) found higher insoluble dietary fibre, protein, fat and ash in intermediated wheat grass compared to annual wheat. Additionally, the higher concentration of carotenoids, hydroxycinnamic and associated antioxidant activity may affect a delayed induction of oxidation, which affects a longer life period in the shelf (Tyl and Ismael, 2019).

Trade-offs

In contrast to annual grain crops, Perennial cereal tend to have lower yields due to a lower harvest index (Culman et al., 2013), and these yields can decline over time (Culman et al, 2013; Hayes et al., 2017; Jungers et al., 2017; Tautges et al., 2018). In the first year, low growth rates increase the risk for weed impacts (Jungers et al., 2018), which might increase application of pesticides (Duchene et al., 2019).

Perennial grain crops have a different nutritional profile to cereal crops and depending on the use, this can have negative impacts for utilisation within for example food production systems. For example, in contrast to annual wheat, intermediated wheat grass shows a deficiency in high molecular weight glutenin subunits and starch, which will make the use for producing dough-based products more challenging (Tyl and Ismael, 2019).

Pasture cropping

Co benefits:

The potential environmental benefits of this system are varied and include increasing soil cover, reducing erosive processes, improving soil structure and organic matter, increasing infiltration and water retention, reducing N lixiviation and sequestering CO_2 (Descheemaeker et al., 2014; Lawes et al., 2014). Regardless of environmental conditions, pasture cropping can have a significant suppression effect on the total weed density and number of weed species. This suppressive effect could be related to early competition for light as a consequence of pasture soil coverage.

Trade-offs:

The economic profitability of pasture cropping systems is conditioned by the coexistence of both cereal crop and pasture without competition penalizing the crop yields. Pasture cropping would negatively affect the main crop yield when the available resources, especially water, are low (Luna et al., 2020).

Liming

Co-benefits:

Liming has been reported to increase the productivity of grasslands (Abdalla et al., 2022). Unlike N fertiliser, which aims to increase grass yield by adding mineral N, liming aims to do so by optimising nutrient availability and plant growth conditions. Thus, correcting soil pH through liming provides the right environment for grassland to reach its growth potential. This reduces the need for animal supplementary feeding and improves the efficiency and sustainability of grazing livestock yields.

In addition, liming enhances soil nitrification (Meiwes, 1995) and thereby, increases soil nitrate concentration and N availability for grass uptake (Fuentes et al., 2006; Holland et al., 2018). It improves soil structure, mitigates soil degradation through its buffering capacity (Keiblinger & Kral, 2018), increases earthworm activities. and makes grass more palatable to animals (DAERA, 2021). The use of high Mg lime for soil treatment increases both soil pH and Mg contents in the soils and thereby, reduces the risks of livestock hypomagnesaemia (i.e. abnormally low magnesium levels in their blood) (Bide et al., 2021).

Although liming does not constitute an agroecological intervention, in croplands and grasslands this practice can contribute to offsetting the negative impact of conventional agriculture on soil health and crop yields. Therefore, we include liming in this review as it can support, over a limited time period, the implementation of other innovative practices, stabilizing productivity both in cropland and grassland systems (Abdalla et al., 2022).Organic inputs

Retention of straw

Co-benefits:

Used as surface mulch, crop residues improve soil chemical and structural properties by increasing SOM concentration (Lal, 2004; Liu et al., 2014). Although the nutrients are only a small portion of straw, returning straw into soil can potentially improve the nutrient status of agricultural soils (Lal, 2004b), which in turn stimulates SOC sequestration due to increased crop rhizodeposition (Kuzyakov & Schneckenberger, 2004).

Stability of aggregates is positively correlated with SOM concentration (Bossuyt et al., 2005). Crop residues, upon decomposition can provide temporary, transient, and persistent binding organic agents (Tisdall & Oades, 1982). There are three mechanisms (physical, chemical, and biological) by which mulching with crop residues stabilizes soil aggregates. Physically, crop residue mulch insulates the soil surface, intercepts the raindrops impacting the soil surface, and moderates freezing-thawing and wetting-drying cycles of surface soil (Blanco-Canqui, 2009). Chemically, it releases substances and compounds such as polysaccharides, humic compounds, and organic mucilages, which enmesh and glue the primary and secondary soil particles into stable aggregates. Biologically, it stimulates activity of macro (e.g. earthworms) and microorganisms (e.g. fungi) to promote formation and stabilization of aggregate.

Crop residue mulch can also serve as a natural blanket to protect the soil surface against the erosive impacts of rainfall and blowing wind. It buffers the soil surface from excessive compaction, surface sealing, and crusting while reducing the breakdown and dispersion of soil aggregates. Maintaining a complete and continuous cover with crop residue mulch on the soil surface is essential to reduce formation of surface seals (Ruan et al., 2001). Cassel et al. (1995) reported that tillage practices such as no-till, which leave crop residues on the soil surface, eliminate surface sealing and crusting.

Trade-offs:

The effect of straw incorporation on subsequent crops is inconclusive and has been shown to vary with system, location and climate (Coulter & Nafziger, 2008; Malhi & Lemke, 2007). Borresen (1999) reported significantly increased wheat grain and straw yields as a result of straw incorporation (Bakht et al., 2009; Børresen, 1999). In addition, Van Den Bossche et al. (2009) showed that straw incorporation enhances soil fertility and may reduce fertiliser requirements. By contrast, Christian et al. (1999) reported lower wheat grain yields, and Coulter and Nafziger (2008) found increased fertiliser N requirement when straw was incorporated. Furthermore, the use of tillage systems that incorporate straw below the seed have been shown to reduce plant height and rate of wheat plant development (Wuest et al., 2000).

Organic manure

Co-benefits:

The meta-analysis of Sanden et al. (2018) reported that organic fertilization reduces bulk density, run-off and sediment yields by 4%, 22% and 16% respectively. The spreading of FYM, slurry, compost and crop residue incorporation can increase soil N contents by 10%, 13%, 14% and 2%, respectively. Compost amendments can increase soil pH by 7%, and FYM amendments increase C/N ratio by 11%.

Regarding soil biological quality, earthworm number, earthworm biomass and microbial populations are positively affected by organic fertilization. The amendment of cattle slurry, for instance, can increase earthworm number, earthworm biomass, microbial biomass C, and bacterial phospholipid-derived fatty acids, whereas, compost amendments was found to significantly increase only earthworm biomass.

Trade-offs:

With organic fertilisation it is difficult to optimise N availability in soils to match plant N demands (Sanchez et al. 2004). Therefore, a potential trade-off is the loss of nutrients through runoff and leaching soil processes.

5.3 Diverse crop rotation

Cover crop

Co-benefits:

Cover crops can increase water holding capacity, soil porosity, aggregate stability, the size of the microbial population and its activity and nutrient cycling (Haruna & Nkongolo, 2015; Lotter et al., 2003). In particular, the non-legume cover crops can absorb excess nitrate from the soil, increase crop biomass, and improve soil quality (White et al., 2016).

Additional co-benefits from cover crops can derive from their capacity to reduce soil erosion, improve water quality, enhance biodiversity, and increase of reflected radiation at soil surface. In this context, legume cover crops have great potential to improve biodiversity in agroecosystems, and serve as break-crops which help to reduce weeds, pest and disease associated with short rotations (Voisin et al., 2013). Grain legume crops may also increase yields of subsequent cereals by approximately 29% (Cernay et al., 2018). However, as reported in Section 4 the meta-analysis of Sanden et al. (2018) showed that yields increased by only 4% when catch crops were included in the crop rotation.

Most crop production guides recommend that farmers reduce inorganic fertilizer inputs to crops that follow legume cover crops (Clark, 2007). Typical recommendations for non-legume cover crops are to maintain the same fertilization levels as with no-cover crop fallows. However, on decadal timescales SOM content increases for both legume and non-legume cover crops resulting in lower fertilizer recommendation rates. Finally, an additional co-benefit from legume cover crops it that they can be used as a co-product for livestock feed (Voisin et al., 2013).

Intercropping

Co-benefits:

Management of cropping systems using intercropping is recognised to increase on-farm diversity enhancing the capacity of agroecosystems to maintain soil fertility, regulates water use, reduces the pressure of pests and diseases, and upholds better yield (Malézieux et al., 2009; Scherr & Mcneely, 2008). In addition, intercropping can be an important strategy in conventional agriculture to promote functional biodiversity by increasing species richness (Connolly et al., 2001), and by enhancing the functioning of ecosystems processes via altered or improved resource acquisition, partitioning and niche differentiation (Moonen & Barberi, 2008). Intercropping also help to increase aboveground and belowground biodiversity, improving agroecosystem regulation (Mace et al., 2012). Multiple cropping systems can regulate pests by preventing their growth, reproduction, or dispersal. Pest dispersal and reproduction can be reduced by resource dilution and spatial disruption. The habitat fragmentation can prevent the spread of disease or make it difficult for insects to find food or mating sites and has been shown to be efficient at control-ling airborne diseases (Gaba et al., 2015).

Trade-off:

Multiple cropping systems may foster populations of undesirable organisms (e.g., slugs in cover crops) which can have an impact on productivity (Gaba et al., 2015).

Legume crops included in rotation

Co-benefits:

Incorporating legumes into typical cereal rotations can bring benefits in terms of reducing environmental impact across multiple crops and derived products. Legume in rotations have the potential to reduce GHG emissions (especially from fertiliser production and use), acidification, terrestrial and aquatic ecotoxicity burdens, among others (Nemecek et al., 2008).

Trade-offs:

A possible trade-off of legume cultivation is higher rates of nitrate leaching (Nemecek et al., 2008; Watson et al., 2017). The potential for SOC accumulation depends on the quality and quantity of residues (Watson et al., 2017). Even though legumes are known to produce more N-rich residues compared with cereals (Begum et al., 2014; Laudicina et al., 2014), they typically produce less residue biomass (Begum et al., 2014). Some studies point to a SOC decrease when a legume crop is introduced into cereal-dominated systems, due to the smaller amount of above and below ground biomass generated by legumes when compared with cereals (Costa et al., 2020).

Lay-arable

Co-benefits:

Improvement of soil fertility by legume-based pasture grassland is an important benefit for arable crops, with annual biological N fixation of 100–300 kg N ha^{-1} (Lemaire et al., 2015). A number of studies showed how growing crops after grassland reduces the response to N fertilizer application and still provides greater yield than continuously grown crops with sufficient N fertilizer (Lemaire et al., 2015).

Introduction of grasslands into crop rotations also offers opportunity to better control weeds and reduce herbicide applications through integrated weed management (Entz et al., 2002; Nazarko et al., 2005; Ominski et al., 1999). Including perennial grasses or forage legumes-like alfalfa in crop rotations disrupts the lifecycle of weed species adapted to annual cropping and reduces their proliferation during subsequent crop phases (Andersson & Milberg, 1998; Katsvairo et al., 2006).

Grass sods planted across an entire field or simply in strips within a field can reduce water runoff and soil loss by promoting water infiltration and trapping sediment. Finally, the perennial habitats represented by patches of grassland may have strong impacts on the population and community dynamics of various soil organisms (Lemaire et al., 2015).

Trade-offs:

In grassland–crop rotations, the termination of grassland with tillage practice can induce a large flux of N mineralization which presents the potential for nitrate leaching risk (Eriksen & Jensen, 2001; Vertes et al. 2007). This is particularly true when the subsequent crop is not able to develop an efficient root system rapidly enough. The rotation management in grass-crop rotation offers opportunities to reduce nitrate leaching by ploughing the grassland in spring, shortly before crop sowing (Eriksen, 2001; Eriksen et al., 2008).

5.4 Multifunctional land-use systems

Silvopasture

Co-benefits:

Silvopasture can increase forage productivity and quality (Jose & Dollinger, 2019), with the forages and trees providing food and shelter for a variety of wildlife, boosting plant, animal, and invertebrate biodiversity (Jose & Dollinger, 2019; Mcadam & Mcevoy, 2009; Pent & Fike, 2021). A study conducted in North-East Scotland found tree biomass is significantly greater in silvopature systems *vs.* woodland (Beckert et al., 2016).

In addition, silvopasture systems have a positive impact on water quantity and quality versus traditional livestock systems (Jordon et al., 2020; Pent & Fike, 2021). Converting forest to silvopasture may reduce evapotranspiration, reducing the impact of drought periods and increasing stem growth (Coble et al., 2020; Stewart et al., 2020).

Trade offs

High intensity grazing of a silvopasture system may be problematic, leading to increased soil penetration resistance and decreased infiltration (Mayerfeld et al. 2021). Furthermore, field-edge forestry systems such as riparian strips or shelter belts are more effective than silvopasture in mitigating flood risk (Jordon et al., 2020). Overall, the influence of silvopasture on soil quality indicators is still unclear (Jose & Dollinger, 2019), and as with all livestock systems there is a risk of soil compaction (Pent & Fike, 2021).

Silvoarable

Co-benefits:

Compared to reforested land, silvoarable systems showed a tendency towards higher diversity. An analysis on different taxonomic groups showed that birds and arthropods are significantly more diverse across all agroforestry systems. For arthropods a higher resolution was available with subgroups on different taxonomic levels, such as bees or spiders (Mupepele et al., 2021). Staton et al. (2019) reported a reduced arthropod herbivore abundance in silvoarable alleys than in arable control conditions.

Trade-offs:

Silvoarable systems have been found to increase the abundance of pests such as slugs relative to arable controls (Staton et al., 2019). Similar effects have been reported in and adjacent to flower-rich field margins (Eggenschwiler et al., 2013; Frank, 1998), which suggests that areas which provide a refuge from tillage could boost slug populations. The above findings indicate that the natural enemies of pests are more abundant in silvoarable alleys compared with arable systems, although there are no clear differences in responses among natural enemy taxa (Staton et al., 2019).

Vegetated strips

Co-benefits:

Vegetated strips may have a multi-functionality that covers a range of processes, including protection of water quality in surface waters and soil conservation of slopes, habitat improvement, biodiversity, shading, carbon sequestration, flow capture, biomass yields, landscape diversity, and societal services (Stutter et al., 2012). As vegetated strips comprise of a variety of different vegetation types that are managed for different purposes, their effects on biodiversity and associated ecosystem services may vary. For instance, pollinator habitat enhancement in the form of hedgerows and flower-rich strips may contribute to yield on adjacent fields (Hoehn et al., 2008), but also overall biodiversity and biological control potential in the surrounding landscape (Wratten et al., 2012). Vegetated strips established using densely planted perennial grasses may primarily benefit invertebrates for pest suppression (Bianchi et al., 2006), but also increase the availability of suitable nesting sites for ground-foraging farmland birds on adjacent crop fields (Josefsson et al., 2013).

Although in this review we did not include riparian buffers, these semi-natural vegetation systems are thought to be an effective, sustainable means of protecting aquatic ecosystems against anthropogenic inputs of N (Mayer et al., 2007). The extent to which riparian buffers attenuate N and subsequently improve water quality is thought to be a function of buffer width in concert with landscape and hydrogeomorphic characteristics (Vidon & Hill, 2004). In particular, the width of a buffer was estimated to accounts for about 80% of that buffer’s N removal effectiveness (Phillips, 1989).

Trade-offs:

Vegetated strips around and within fields may influence crop yields. Although field margins can support beneficial invertebrates such as natural enemies of pest invertebrates, this practice may harbour weeds, pests and diseases (e.g. viruses), which could potentially create a conflict between crop yields and biodiversity conservation (Capinera, 2005; Marshall & Moonen, 2002; Walton & Isaacs, 2011). Increased habitat heterogeneity may also have negative impacts on some migratory (grass-eating) species (e.g. geese) or farmland species such as skylarks that rely on the cropped area of large fields, for breeding and foraging (Danhardt et al., 2010; Silva et al., 2010).

5.5 Pasture productivity

Rotational grazing

The environmental impact and efficacy of rotational grazing is contested (Briske et al. 2008; Roche et al. 2015; Venter, Cramer and Hawkins 2019), but a global meta-analysis found that its apparent impact on soil health indicators mean it could create opportunities for climate change mitigation (Byrnes et al. 2018).

Co-benefits:

In general, rotational grazing can help improve productivity, weight gain or milk yield per acre, and overall net return to the farm. Rotational grazing allows for better manure distribution that acts as a source of nutrients to the soil. It can help extend the grazing season, allowing a producer to rely less on stored feed and supplement, and has the potential to reduce machinery cost, fuel, supplemental feeding and the amount of forage wasted.

Compared to continuous grazing, rotating grazing promotes increased plant growth and may reduce bare soil (Jacobo et al. 2006; de Otálora et al. 2021). It leads to higher bulk density than non-grazing systems, but lower bulk density than continuous grazing (Byrnes et al. 2018). The lower bulk density can lead to higher water retention capacity, even with significantly higher stocking rates than in continuous grazing (Teutscherova et al. 2018). The use of specific rotational grazing techniques, such as adaptive multi-paddock grazing over continuous grazing approaches may reverse soil degradation resulting from traditional livestock stocking practices (Teague and Kreuter 2020). Using a 'biodiversity-friendly rotation' system for sheep and cows can increase abundance and species-richness of flower visiting insects (Enri et al. 2017)

Trade offs

Plant yields appear to be equal or greater in continuous grazing systems (Briske et al. 2008), and rotating grazing can result in higher N and C runoff (Pilon et al. 2019). Moreover, rotating grazing on its own does not reduce P runoff *vs.* continuous grazing (Anderson et al. 2019).

Multispecies leys

Co-benefits:

Compared to simple swards, the advantages of multispecies lays are often associated with more efficient use of resources. They provide a positive strategy to support functional biodiversity at field, farm and landscape levels. The increased on-farm diversity also enhances the capacity of agroecosystems to maintain soil fertility, and regulates water flow.

Multispecies leys have the potential to reduce N losses from livestock (Beukes et al., 2014). N intake has been shown to have a positive linear relationship with urine, faecal, and N excretion (Mulligan et al., 2004) (Mulligan et al., 2004). The inclusion of herb species such as chicory (*Cichorium intybus* L.) and ribwort plantain (*Plantago lanceolata* L.) in multispecies grazing swards has been shown to reduce urinary N excretion (Bryant et al., 2017; Dodd et al., 2018). Additional benefits from multispecies leys might be derived from the improvement of livestock products (beef and milk) (Mccarthy et al., 2020).

Trade-off:

The species diversity can cause problems if new species or varieties are introduced in places where they are not grown historically (Müller, 2015).

5.6 Extensive livestock

Extensive grazing (reduced stocking density)

Co-benefits

The evidence supporting grazing management has been mixed, as it has been found to increase (Guo et al., 2017), decrease, or have no effect on plant biomass (Zhan et al., 2020). Large differences are driven by different grazing patterns, and when compared with light grazing and high grazing , medium grazing may increase below-ground biomass , as medium grazing contributes to higher species richness than light or high grazing (Gong et al., 2014).

The effects of grazing intensity on soil properties is unclear in the literature, particularly the correlation between soil nutrient and microbial activity at a large scale. Insights about the linkages of grazing intensity and soil nutrients with soil microbial changes have been limited to individual studies (Ford et al., 2013; Strickland & Rousk, 2010). However, depending by the habitat, the extensification of grazing has been found to reduce the risk of wildfires, as well as the need for imports of alternative feeds (Ruiz-Mirazo 2011).

Given the above uncertainties, here we hypothesise that the environmental effects of extensification of grazing are, in general, the opposite (i.e. positive) of those caused by intensive grazing. Positive effects include improvements in aboveground community biomass (Yan et al., 2013; Zhang et al., 2015). In particular, the increase in aboveground biomass has a direct positive effect on litter input into soil, as well as soil C and N availability (McSherry & Ritchie, 2013; Xiong et al., 2016; Yan et al., 2013) (McSherry and Ritchie, 2013; Yan et al., 2013; Xiong et al., 2016). Furthermore, extensive grazing possibly increases root biomass due to more aboveground photosynthate allocation to belowground, and induces higher plant coverage that control soil temperature and wind erosion (Tian et al., 2016). This soil protection reduces soil water losses and its availability in grasslands (Wang et al., 2012). In addition, extensive grazing limit topsoil compaction due to the reduced livestock trampling (Tang et al., 2019).

Trade offs

Decreasing stocking rates, while maintaining overall herd size will require more land leading to either adoption of other measures (e.g., silvopasture, rotational grazing), or less land available for other farm activities. Modern intensive livestock yield systems, particularly beef, require significantly less land than extensive systems (Swain et al. 2018). The quality and quantity of pasture growth tends to increase with increasing stocking rates (MacDonald et al. 2008).

Grass-based livestock diets

Co-benefits:

As highlighted in Section 3, grass-fed livestock may have a number of environmental and human health co-benefits. Grass-fed beef has higher micronutrient concentrations and a fatty acid profile that might lead to improved human health outcomes relative to consumption of grain-fed beef (Daley et al 2010). Furthermore, grass-fed beef may promote food security in cropland-scarce regions because it can be grown on land not suitable for crop production (Smith et al 2013).

Reducing dietary crude protein and ruminally degradable protein concentration can reduce ammonia emissions from manure, through a marked reduction of urinary urea excretion, ammonia concentration and potentially N_2O emissions from dairy manure (Lee et al., 2012; Schils et al., 2013). Shifting N excretions from urine to faeces is expected to reduce N_2O emissions from manure application because of the lower concentration of available N in manure, depending on manure storage time and conditions (Hristov, Oh, et al., 2013). In this context, tanniferous forages can be used for this purpose and have been shown to reduce urinary N as proportion of total N losses by approx. 9% (Carulla et al., 2005) and 25% (Misselbrook et al., 2005).

Trade-offs:

Grass-fed livestock has been reported to have higher land use requirements and GHG emissions than grain-fed (Clark & Tilman, 2017; Smith et al., 2008). These two trade-offs derive from the lower macronutrient densities and digestibility of feeds used in grass-fed systems (AFRIS-FAO, 2016), and because they cause grass-fed livestock such as beef to require higher feed inputs per unit of beef produced than grain-fed systems. In addition, lifetime CH_4 emissions, and thus GHGs per unit of food, tend to be higher for grass-fed animals as they grow slower and are slaughtered 6–12 months older than grain-fed animals (Clark & Tilman, 2017).

Conversion of temporary leys to permanent pasture

Co-benefits:

Permanent pastures provide a number of benefits such as soil health and ecosystem services (Eyles et al. 2015). In particular, the prevention of erosion through maintenance of groundcover and the adoption of managements promoting deep C sequestration are likely to support the increase SOC in pasture soils over a decade or longer (Eyles et al. 2015). Silburn et al. (2007) found that pasture systems are inherently more efficient in using rainfall which results in less runoff and deep drainage than cropping systems. Specially, in areas of high salinity risk, they found that pastures have an important role in reducing deep drainage. A study by Norton et al. (2020) investigated high proportion of legumes in swards and found that this is closely linked to greater applied phosphorus in combination with high stocking rates and lime applications.

Species rich Permanent grasslands such as hay meadows are also an important habitat for insect communities, including pollinator populations which are in pan-European decline (Scohier et al 2012). Rotational grazing plays a key role in the preservation of flower-rich patches and the maintenance of sward heterogeneity. Scohier et al. (2012) found benefits of rotational grazing on permanent grasslands mainly on bumblebee density and species richness, with some additional effects on local density of butterflies.

Trade-offs:

Lambert et al. (2004) concluded that when considering pasture renewal, it is important to consider a cost-benefit analyses, selecting the options that best meet the needs in establishment and subsequent management practices.

6 Impact of the agroecological practices on agriculture: an overview

Most of the agroecological practices reviewed here have only a low or medium implementation level in current UK agriculture (Table 6). This is due to still-limited scientific knowledge or low practical on-farm experience with practices such intercropping, pasture cropping, multispecies leys (Wezel et al., 2014). With agroforestry systems there is a time delay before yields and ecological benefits are achieved, and arable systems (e.g. such as silvoarable) are likely to require significant investment, training and re-design of the farm business. Table 6 outlines the number of indicators positively or negatively affected by the AEFPs, as well as the number of agroecological principles that are addressed. In general, a farming system that scores high in agroecological principles can achieve a higher level of sustainability *via* agroecological transformation (Notenbaert et al., 2021). This means that, AEFPs with a low level of integration such as intercropping, multispecies leys or silvopasture have the potential to generate win-win situations from the production of several ecosystem services, achieving the sustainable intensification of yields within the farms. A number of AEFPs such as rotational cropping are already well integrated in UK's agriculture, indicating that there are opportunities to introduce agroecology with relatively low investments and no change in land-use. It is important to point out that the trade-offs outlined in Table 6 will not be experienced on every farm setting. Within the variability of the farm context, in the agroecological transition farmers and agronomists might be able to work to avoid or minimise trade-offs and maximise co-benefits.

Table 6: Summary of the number of co-benefits (positive effects) and trade-offs (negative effects) generated from the implementation of different agroecological farm practices (AEFP) across arable (A), livestock (L) and mixed (M) farming systems. For the indicator GHG emissions, C stocks and Yield benefits and trade-offs are expressed with circle symbols. Green, red and grey circles correspond to positive, negative and no effect, respectively. Empty cells indicate that insufficient information was available. More than one coloured circle reflects contrasting findings in the literature. The category co-benefits show the total number of indicators with only positive effects. The category AEP summarise the agroecological principles addressed by each AEFP. The category Integration represents the level of use of each AEFP in today's agriculture, based on scientific knowledge and practical on-farm experience with practices.

System	AEFP	GHG	C stocks	Production	Co-benefits	AEP addressed	Integration
A	No-till - Minimum tillage	red	green	red	5	3	High
A	Perennial cereal crops		grey	red	11	5	Low
M-L	Pasture cropping		green	grey	8	3	Low
A	Liming*		green	green	5	3	High
A	Retention of straw	red	green	red, green	11	3	High
A-M-L	Organic manure	red	green	grey	5	2	Medium
A	Cover crop	red	green	grey	17	6	Medium
A	Intercropping			green	12	5	Low
A-M	Legume crops		grey		11	6	Medium
A	Ley-arable		green	grey	11	5	Low
A	Silvoarable	grey	green	red, grey	6	2	Low
M-L	Silvopasture		green	green	10	4	Low
A-M-L	Vegetated strips	green	green	green	17	5	Low
M-L	Rotational grazing		green	green	12	6	High
M-L	Multispecies leys		green	green	12	5	Low
M-L	Extensive grazing	red	green	grey	9	4	Low
M-L	Grass-based diets	red, green, grey	green		2	1	Low
M-L	Permanent pasture		green	red	10	3	Medium

Below we summarise the co-benefits described in Sections 4 and 5 and outlined in Table 5. For each AEFP we report the evidence found in literature into two sub-sections: 1) GHG emission, C stocks and productivity, and 2) additional co-benefits.

6.1 Reduced soil disturbance

No-till and minimum tillage

Reduced tillage systems are primarily based on reducing soil disturbance by restricting land preparation activities to a shallow depth and eliminating soil inversion, while conserving and managing crop residues. Overall, we found that these practices can generate a relatively low level of agroecological transition at farm level when applied on their own (Table 6).

GHG emissions, soil C stocks and productivity:

The implementation of minimum tillage has no effect on CO_2 and N_2O emissions, but no-till can have a negative impact on N_2O emissions. We found that only minimum tillage has a strong positive effect on increasing soil C stock, while no-till has strong negative impact on crop yields. The trade-off of no-till on production could be explained by the negative effect that no-till has on soil indicators such as bulk density, soil compaction and soil erosion / run-off processes.

Additional co-benefits:

Both no-till and minimum tillage are recommended for soil and water conservation, and have been reported to significantly increase available P, as well as earthworm number and biomass. The low positive resilience effect deriving from these practices is connect to the improvement of a number of soil health indicators (e.g. water holding, soil structure, and soil biodiversity).

Perennial cereal crops

Overall, we found that perennial cereals can deliver up to 11 co-benefits spread across 5 agroecological principles.

GHG emissions, soil C stocks and productivity:

In contrast to annual grain crops, perennial crops tend to have lower yields. Importantly, low growth rates may increase the risk of weed impacts and the need to increase weed control activities. Compared to annual crops perennial cereal might have a positive effect on soil C stock (Ledo et al., 2020). However, a study carried out in New South Wales, Australia harvestable yield found significant declined of yield after the first year. We did not find any relevant information on the effect of perennial cereal on GHG emissions.

Additional co-benefits:

The perennial nature of these systems means that there is surface cover during the entire year and a more persistent root system. Both of which bring a strong reduction in soil erosion. Additionally, the well-developed root system and reduced disturbances improve the soil health by improving the porosity, increase the microbial activity and improve the aggregate stability; thus increasing the resilience of soil to climate change stressors.

Nutrient holding capacity is increased in perennial grain crop systems, compared to annual crops. This shows potential for an improved nutrient use efficiency and reduced N leaching, which ultimately impact water quality at farm level.

Pasture cropping

GHG emissions, soil C stocks and productivity:

Pasture cropping have positive effects on soil C stocks in arable land. This benefit, in particular, is related to the capacity of this AEFP to improve soil health. We found no evidence on the impact of pasture cropping on GHG emissions, and only weak evidence about a small effect on productivity. In particular, arable crops established in permanent pasture have lower yields than crop monocultures.

Additional co-benefits:

The potential environmental benefits of this system are varied and include increasing soil cover, reducing erosive processes, improving soil structure and increasing infiltration and water retention. Regardless of environmental conditions, pasture cropping can have a significant suppression effect on the total weed density and number of weed species.

Liming

GHG emissions, soil C stocks and productivity:

Liming can contribute to enhanced soil C stock and crop yields. These positive effects can be explained by the fact that lime reduces soil acidification, improving soil physical conditions, reducing aluminium toxicity, and increases soil P and Mg availability when needed.

No scientific evidence has been found on the impact of liming on GHG emissions.

Additional co-benefits:

Liming does not constitute an agroecological intervention *per se*. However, for time limited period in croplands and grasslands this practice can contribute to offsetting the negative impact of intensive agriculture on soil health. This benefit tends to increase earthworm abundance, and as explained above, also soil nutrient management.

6.2 Organic inputs

Organic manure

In mixed farms, soil amendment with organic manure can contribute to the establishment of the recycling of nutrients produced within the farm boundaries, which is an essential principle of agroecology.

GHG emissions, soil C stocks and productivity:

The amendment of organic manure can directly increase soil C storage by inputs of organic matter as well as indirectly by increasing net primary productivity. We found contrasting evidence on the impact of organic manure on crop yields. In particular, Sanden et al. 2018 and Shang et al. 2021 reported a negative and positive effect of FYM on yield, respectively. Farm yard manure applications are difficult to optimise based on type of amendment, soil nutrient availability and crop demands, and depending on the agronomic conditions they can negatively impact CO_2 and N_2O emissions (Table 4). In general, FYM has a small negative effect on N_2O emissions. Slurry was found to increase both CO_2 and N_2O compared to equivalent inorganic fertiliser, while the use of compost does not impact CO_2 or N_2O emissions, and increases SOC stocks.

Additional co-benefits:

In general, organic manure increases soil health by positively affecting earthworm and microbial populations, increases the soil organic matter and soil C/N ratio, and improves soil bulk density.

Retention of straw

GHG emissions, soil C stocks and productivity:

The return of straw to the soil can potentially improve the nutrient status of agricultural soils, which in turn stimulates SOC sequestration. In addition, this AEFP can significantly increase N_2O emissions. Depending on the crop type and quantity of residues left on the ground, straw retention can have contrasting effect on crop yields. Lower yields were identified in Europe compared to straw removal, but a global meta-analysis found that yields increased when residues were retained.

Additional co-benefits:

The cover of crop residue on the soil can protect the soil surface against insolation and erosive impacts of raindrops and blowing wind. It buffers the soil surface from excessive compaction, surface sealing, and crusting while reducing the breakdown and dispersion of soil aggregates. Biologically, it stimulates activity of macro (e.g., earthworms) and microorganisms (e.g. fungi) to promote formation and stabilization of aggregates.

6.3 Diverse crop rotations

Cover crop

Cover crops are grown for ecological services (e.g., soil cover, nutrient capture, fertility improvement, weed suppression) rather than a harvestable product. We found that this AEFP generates a high level of agroecological transition.

GHG emissions, soil C stocks and productivity:

The review of Sanden et al (2008) found that cover crops can significantly increase N_2O emissions, as well as potentially increasing CO_2. The positive impact on soil conditions can result in a significant positive effect on soil C stocks. Cover crops can increase productivity. However, this effect depends also on the crop type and soil conditions.

Additional co-benefits:

Cover crops can improve several environmental indicators, which increase the overall resilience of farm to climate change. In particular, cover crops can increase water holding capacity, soil porosity, soil aggregate stability, and the size of the soil microbial population. Additional co-benefits from cover crops can derive from their capacity to reduce soil erosion, improve water quality, enhance biodiversity, and increase of reflected radiation at soil surface. Importantly, through the use of cover crops it is possible to reduce inorganic fertilization, and in mixed farms they can be used as a co-product for livestock feed.

Intercropping

Intercropping can be an important strategy in conventional agriculture to promote functional biodiversity by increasing species richness (Connolly et al., 2001), and by enhancing the functioning of ecosystems processes within the farm.

GHG emissions, soil C stocks and productivity:

We did not find any information on the impact of intercropping on GHG emissions or on soil C stocks. Crop yields may increase with intercropping due to the complementarity in use of N sources, or due to the facilitation of interplant N transfer and availability of P.

Additional co-benefits:

Management of cropping systems using intercropping is recognised to enhance the resilience of agroecosystems by increasing on-farm diversity, enhancing the capacity of agroecosystems to maintain soil fertility, regulate water use and reduce the pressure of pests and diseases.

In addition, they also help to increase aboveground and belowground biodiversity, improving agroecosystem regulation (Mace et al., 2012). Multiple cropping systems can also regulate pests by preventing their growth, reproduction, or dispersal. Pest dispersal and reproduction can be reduced by resource dilution and spatial disruption i.e. habitat fragmentation.

Legume crops included in rotation

Cereal–legume intercropping or cover crops are a well-known example of a system based on complementary functioning that optimizes the use of N at field scale over a growing season. For this reason, this practice generates a level of agroecology similar to intercropping and cover crops.

GHG emissions, soil C stocks and productivity:

Legumes in rotations have the potential to indirectly reduce GHG emissions from fertiliser production and use. However, in this review we did not find evidence on the any direct effect of legume crops on GHG emission and productivity. Having legumes intercropped with non-legumes (i.e. cereals) could be a way to promote yield stability (Raseduzzaman & Jensen, 2017). Depending by the rotation system, however, studies found that SOC concentration can increase (Mcdaniel et al., 2014) or have no effect on the yield of subsequent crops (West & Post, 2002).

Additional co-benefits:

Incorporating legumes into typical cereal rotations can bring benefits in terms of reducing environmental impact (i.e. resilience to climate change) across multiple crops and derived products. These crops have the ability to fix atmospheric N which is eventually returned to the soil, leading to a reduction in N fertilization needs. In mixed farms, legumes are also an important source of protein for feed and food.

Ley-arable

In mixed farming, the introduction of grasslands within arable cropping systems has the potential to achieve synergy between productivity and other ecosystem services such biodiversity and soil health.

GHG emissions, soil C stocks and productivity:

We did not find any research papers covering the impact of ley-arable on GHG emissions. Improvement of soil fertility by legume-based pasture grassland is an important benefit for arable crops, and we found a positive strong effect on soil C stocks in ley-arable systems. In addition, we found that the ley presence has no effect on the yield of subsequent crops.

Additional co-benefits:

As for intercropping, the rotation of crops with grass pastures can be effective in improving soil structure and fertility, can increase the population and community of various soil organisms, and disrupt pest and disease lifecycles. Introduction of grass into crop rotations also offers opportunity to better control weeds and reduce herbicide applications through integrated weed management. Grass sods planted across an entire field or simply in strips within a field can reduce water runoff and soil loss by promoting water infiltration and trapping sediment. Finally, the perennial habitats represented by patches of grassland may have strong impacts on the population and community dynamics of various soil organisms (Lemaire et al., 2015).

6.4 Multifunctional land-use systems

Silvoarable and Silvopasture

GHG emissions, soil C stocks and productivity:

We found only weak information on the impact of multifunctional systems on GHG emissions. In that respect, silvoarable systems do not impact emissions of N_2O and CO_2. Both silvoarable and silvopasture systems have the potential to increase soil C stocks compared with equivalent mono-cropped land. However, further research in silvoarable systems is needed to confirm this finding. Regarding productivity, we found contrasting results for these two systems, with silvoarable and silvopasture showing low negative and positive impacts on productivity, respectively (Table 5). However, individual studies found that temperate silvoarable systems have the potential to increase productivity (timber) compared with equivalent mono cropped land (Graves et al., 2010); (Gruenewald et al., 2007). Silvopasture can increase forage productivity and quality (Jose & Dollinger, 2019).

Additional co-benefits:

Silvoarable and silvopasture systems are recognised to enhance several environmental indicators which have a positive synergistic effect on the resilience of agroecosystems (Table 5). Silvopasture systems have a positive impact on water quantity and quality versus traditional livestock systems. Compared to reforested land, silvoarable systems showed a tendency towards higher biodiversity. In particular, for arthropods a higher resolution was available with subgroups on different taxonomic levels, such as bees or spiders. It is still debated in the literature whether it is more beneficial to integrate diverse land uses on the same piece of land or keep them separate, particularly in the context of food production and biodiversity conservation.

Vegetated strips

GHG emissions, soil C stocks and productivity:

Overall, we found that vegetated strips can reduce N_2O emission, enhance soil C stocks, and increase crop and livestock productivity at farm level.

Additional co-benefits:

Vegetated strips comprise a variety of different vegetation types that are managed for different purposes. Both hedgerows and grass strips have been reported to deliver a number of ecosystem services, for example by carbon sequestration and N and P interception from water flows, which can enhance the resilience of farms. Depending on the situation and application, these AEFPs can provide protection for topsoil, livestock, and crops, reduce inputs of energy and chemicals, increase water use efficiency, improve air and water quality, sequester carbon, and enhance biodiversity. Vegetated strips established using densely planted perennial grasses may primarily benefit invertebrates for pest suppression, and increase the availability of suitable nesting sites for ground-foraging farmland birds.

6.5 Pasture productivity

Rotational grazing

GHG emissions, soil C stocks and productivity:

No evidence was found on the effect of rotational grazing on GHG emissions. Depending on the stocking rates, rotational grazing may generate win-win relationships at farm level. In particular, we found that rotational grazing can improve soil C stocks, plant productivity in pastures, and livestock weight gain. These effects are driven by the approach characterising this AEFP, which allows the grazed paddock a rest period that permits forages to regrowth, renew carbohydrate stores, improve yield and persistence that extends the grazing season.

Additional co-benefits:

Rotational grazing generates a high level of agroecological benefits generating up to 12 co-benefits spread across 6 agroecological principles. In mixed livestock farming it allows for better manure distribution that acts as a source of nutrients to the soil. This benefit allows producers to rely less on stored feed, supplements, and manure/slurry storage that reduce the point emission sources within the farm. Lower soil bulk density in rotational grazing can lead to higher water retention capacity, even with significantly higher stocking rates than in continuous grazing. In addition, depending on the agroecosystem conditions, it may reverse soil degradation.

Multispecies leys

GHG emissions, soil C stocks and productivity:

Multispecies leys have positive effects on livestock yields (beef and milk), and enhance soil C stocks. In addition, the inclusion of legumes would increase soil carbon sequestration without affecting N_2O emissions. No evidence was found about the impact of multispecies leys on GHG emissions.

Additional co-benefits:

Multispecies leys have the potential to reduce N losses from livestock. They also provide a positive strategy to support functional biodiversity within the farm enhancing soil fertility and water regulation, which can contribute to improve the resilience of pastureland to climate change stressors.

6.6 Extensive livestock systems

Extensive livestock grazing (reduced stocking density)

GHG emissions, soil C stocks and productivity:

Extensive grazing has the tendency to impact GHG emissions per unit of animal product, and increase above- and below-ground biomass in grasslands. A study from China found that reduced stocking density significantly increases CO_2 emissions and decreases CH_4 emissions compared to high stocking densities (Tang et al. 2019). The increase in biomass yields can have a direct positive effect on litter input into soil, as well as soil carbon. Aboveground carbon stocks in herbage biomass are also increased when stocking density is reduced. Although livestock grazing reduces SOC compared to no grazing in temperate regions, there is no difference between light vs heavy grazing in this effect. Excluding the reduction of stocking rate, no evidence was found concerning a direct effect of extensive grazing on livestock productivity.

Additional co-benefits:

Reduction of stocking density in extensive grazing leads to an improvement in individual animal and total herd productivity. The effects of grazing intensity on soil properties is unclear in the literature, particularly the correlation between soil nutrient and microbial activity at a large scale. However, the extensification of grazing has been found to reduce the risk of wildfires compared to no grazing, as well as the need for imports of alternative feeds.

Grass-based livestock diets

GHG emission, soil C stocks and productivity:

A grass-fed livestock diet has been reported to decrease CO_2 emissions, increase CH_4 emissions, and have no impact on N_2O emissions. No evidence was found on the impact of grass-based diets on livestock productivity. However, livestock growth rates are slower on grass-based diet than cereal-based.

Additional co-benefits:

Regarding the indirect impact of grass-based livestock diets, it is plausible to assume that the conversion to grassland of cropland previously used to grow cereals fed to livestock would increase soil carbon in the land affected by LUC. In addition, individual studies showed that grass-based diets can improve food security as it involves land not suitable for crop yields, and mitigation of manure emissions (Lee et al., 2012; Schils et al., 2013, Hristov, Oh, et al., 2013). In particular, reducing dietary crude protein and ruminally degradable protein concentration can reduce ammonia emissions from manure. The mitigation effectiveness of grass-dietary strategies differs across different types of ruminants. Feeding more highly digestible grass seems to be more effective for dairy and beef cattle than for sheep.

Conversion of temporary leys to permanent pasture

GHG emission, soil C stocks and productivity:

No evidence was found on the impact of converting temporary leys to permanent pastures on GHG emissions, and only weak evidence was found on a potential decrease of productivity. However, the review of Smith et al (2010) showed strong evidence for an increase of soil C stock in permanent pastures.

Additional co-benefits:

Permanent pastures provide a number of benefits such as soil health, prevention of erosion through maintenance of groundcover, less runoff and deep drainage compared cropping systems. The synergistic effect of these positive indicators enhance the resilience of livestock farms to climate change. In areas of high salinity risk, pastures have an important role in reducing deep drainage. Permanent grasslands are also an important habitat for insect communities, including pollinator populations. In permanent pastures, rotational grazing plays a key role in the preservation of flower-rich patches and the maintenance of sward heterogeneity, which has been reported to increase bumblebee density and species richness.

7 Integrating agroecology in the CCC Balanced Pathway Scenario to reach net zero

In the Balanced Pathway Scenario the CCC included six actions aiming to support the sustainable transition of the agricultural sector towards net zero by 2050 (CCC, 2020a). These actions include measures to release land - shifting people’s diet towards lower levels of animal products, reducing food waste across the supply chain and improvements in agricultural productivity - low carbon farming practices, low carbon machinery, land use change to increase woodland cover, energy crops and restore degraded peatlands. Through the Balanced Pathway Scenario, therefore, CCC set out a narrative of net zero based on the increase of productivity and efficiency across AFOLU, which help deliver emission reductions and carbon sequestration while sustaining the ecosystem services that agriculture depends on. Agricultural efficiency will be achieved through a combination of change in farming practices and innovation. Support for research and development will be required to determine high yielding varieties, livestock breeding for beneficial traits, high-tech solutions to improve usage of external inputs (e.g. drones or GPS directed robots), and low-carbon electricity or hydrogen agricultural machineries. However, based on the current knowledge on ecological genetics and engineering techniques for such sophisticated interventions, such products are not expected to become part of conventional agriculture for quite some time (Hilbeck and Oehen, 2015).

For agroecological approaches to contribute effectively to the sustainable transition towards net zero requires radical move towards a new type of agri-food economy, based on market mechanisms and organisations interwoven with active farmers and consumer’s participation (Horlings and Madsen, 2011). In contrast to the productivist approach, which relies on intensification based on external inputs and technology, agroecology targets diversification in agricultural approaches along with decentralization of industrial agriculture. This means that, for agroecology the sustainable transformation of conventional agriculture should consider not only the protection of productivity in terms of yields, but also the possibility to increase food sovereignty (Herren, 2015). By applying agro-ecological practices farmers do more than just produce food. They also protect biodiversity and associated ecosystem services. In addition, the development of regional markets based on localised food production and consumption ensures a decent livelihood for small-scale farmers and a fair food price for consumers. This in turn makes farming economically more competitive and resilient to global contingencies, supporting a fairer transition to net zero.

In Table 7, we highlight where the Balanced Pathway Scenario would allow the implementation of agroecological practices. In particular, we assume that a number of AEFPs can be implemented in the measures that aim to reduce or capture emissions such as low carbon farming through novel livestock diets, livestock health, soil measures, silvoarable or silvopasture, and hedge planting (1C, 1D, 1E, 4O, 4P), or linked to making changes to society towards greater sustainability adopting options to release agricultural land through higher agricultural productivity and indoor agriculture (2H, 2I). Below, we discuss the agroecological reasons and environmental benefits from these AEFPs.

Table 7: Summary of the core actions and measures of the Balanced Pathway Scenario (CCC 2020a), and the reviewed agroecological farm practices (AEFP) that can contribute toward Net Zero in the agriculture and land use sectors. Although not mentioned in the Balanced Scenario, in measure E -Soil measures we included AEFPs [(a)] that can contribute to enhance soil organic matter, soil structure and limit soil erosion.

<table>
<tr><th colspan="2">Action</th><th colspan="3">Measures</th><th>CCC Balanced Pathway</th><th>AEFP</th></tr>
<tr><td rowspan="8">1</td><td rowspan="8">Low carbon farming</td><td>A</td><td colspan="2">Breeding measures</td><td>Select beneficial traits to lower emissions intensity</td><td></td></tr>
<tr><td>B</td><td colspan="2">Increasing the milking frequency</td><td>Increase milk yield by 10%</td><td></td></tr>
<tr><td>C</td><td colspan="2">Livestock diets</td><td>High sugar content grasses.
Grass-based.
High starch diet for dairy cattle.</td><td>Grass-based livestock diets, Multispecies leys, Rotational grazing, Ley-arable.</td></tr>
<tr><td>D</td><td colspan="2">Livestock health</td><td>Change housing and management.
Improve prophylaxis.</td><td>Grass-based livestock diets, Extensive livestock grazing, From temporary leys to permanent pasture, Ley-arable, Silvopasture</td></tr>
<tr><td>E</td><td colspan="2">Soil measures</td><td>Soil measures include approaches such as grass and legume mixes, cover crops and grass leys.</td><td>Legume crops in rotation, Cover crop, Ley- arable, Grass-based livestock diets, Intercropping, Multispecies leys, Temporary leys to permanent pasture, (No-till / minimum tillage, residue retention, organic manures, perennial cereal crops and pasture cropping) [a]</td></tr>
<tr><td rowspan="2">F</td><td rowspan="2">Manure management</td><td>Anaerobic digestion</td><td rowspan="2">Anaerobic digestion
All slurry tanks and lagoons are fitted with an impermeable cover.</td><td rowspan="2"></td></tr>
<tr><td>Covering slurry tanks</td></tr>
<tr><td>G</td><td colspan="2">Low carbon farm machinery</td><td>On-farm take up of biofuels, electrification, heat pumps and hydrogen.</td><td></td></tr>
<tr><td rowspan="4">2</td><td rowspan="4">Options to release agricultural land</td><td>H</td><td colspan="2">Higher agricultural productivity
(free up 0.5-0.6 mil. ha)</td><td>Increase farm water storage capacity.
Adopt farm practices to improve soil fertility and structure
Increase livestock utilisation rates by 30% with paddock grazing.</td><td>Intercropping, Rotational grazing, Multispecies leys</td></tr>
<tr><td>I</td><td colspan="2">Indoor horticulture (free up 7000 ha)</td><td>Shift 10-50% of horticulture production indoors in controlled environment</td><td></td></tr>
<tr><td>L</td><td colspan="2">Healthier human diet (free up 3 mil. ha)</td><td>20-40% reduction in meat and dairy consumption</td><td></td></tr>
<tr><td>M</td><td colspan="2">Food waste reduction</td><td>50% reduction of food waste by 2030, 60% by 2050</td><td></td></tr>
<tr><td>3</td><td>Forestry and woodlands</td><td>N</td><td colspan="2">Afforestation and sustainable management</td><td>80% of broadleaf woodland area is under sustainable management by 2030
Woodland cover increases by 17% and 20% by 2050</td><td></td></tr>
<tr><td rowspan="2">4</td><td rowspan="2">Agroforestry and Hedges</td><td>O</td><td colspan="2">Agroforestry</td><td>Developed on 5-15% of agricultural land by 2050</td><td>Silvoarable, Silvopasture</td></tr>
<tr><td>P</td><td colspan="2">Hedgerows</td><td>40% increase in hedge length by 2050</td><td>Vegetated strips: hedgerows, grass margins, tree strips</td></tr>
<tr><td>5</td><td>Peatlands</td><td>Q</td><td colspan="2">Restoration and sustainable management of peatlands</td><td>Restore 79% of peatland by 2050
35% of lowland cropland sustainably managed</td><td>Cover crops, permanent pasture</td></tr>
<tr><td>6</td><td>Bioenergy</td><td>R</td><td colspan="2">Miscanthus, short rotation coppice, short rotation forestry</td><td>Increase area to 0.7 million ha by 2050</td><td></td></tr>
</table>

7.1 Low carbon farming (Action 1)

Low carbon farming includes measures that aim to reduce emissions, improve productivity and resource use through farming practices and the adoption of technological options. The CCC estimates that these core measures can reduce GHG emissions from the agricultural sector by 15% by 2050.

Livestock diets and health (Measure 1C, 1D)

Livestock diets (1C) and livestock health (1D) includes measures comprising animal feed and additives that can reduce enteric emissions in cattle and sheep, one that improves the feed conversion efficiency, and preventative measures e.g. changing housing and management to reduce stress and exposure to pathogens, vaccination, and improved screening, and curative treatments such as antiparasitics and antibiotics. In view of these concerns, agroecological approaches such as multispecies-leys and livestock extensification can play an important role in the measures 1C and 1D.

We found that grass-based diets can increase CH_4 emission and reduce growth rates in beef farming. Change in livestock diets can have a profound effect on manure emissions, as it drives the volume and composition of manure. In particular, grass-based diets affect the amount, form and partition of N excretion between urine and faeces, and the amount of fermentable organic matter excreted (Gerber et al., 2013). In particular, reducing crude protein and ruminally degradable protein concentration can positively reduce urinary urea excretion, ammonia concentration and potentially N_2O emissions from dairy manure. However, grass-based low-protein diets for ruminants should be balanced for amino acids to avoid feed-intake depression and decreased production (Hristovet al., 2013).

When compared against intensive grazing, extensive grazing combined with reduced stocking density can address several agroecological principles. These benefits include positive impacts on biodiversity, soil C sequestration, and animal health (Table 4 and 5). The reduction of the number of animals, in particular, allows for the provision of adequate feed to a herd, leading to the improvement of the health in individual animal, increase in total herd productivity, and the reduction of CH_4 emissions for both the total herd and per unit of animal product (Hristov, Ott, et al., 2013). The introduction of extensive farming might generate important trade-offs that need to be managed. Extensive grazing has the tendency to increase GHG emissions per unit of animal production. In addition, the transition to extensive grazing carries the land-requirement concern if the same level of production is required. Ultimately, the role of the livestock sector is related to the bigger question of how protein is produced, the role of public demand for these products, and how much could be achieved adopting sustainable crop-livestock systems that include more encompassing agroecological interventions conceived on the landscape scale.

To generate the ecosystem services provided by extensification, it is important to define a safe operating space in the livestock sector. In the UK, there is large variation between animal genetics (breeds – both late and early maturing, and crosses), grassland resources (upland and lowland-improved pasture) and management type (proportion of grazing to cereal in the finishing/fattening stage) (Magowan et al., 2020). Approximately half of the beef produced in the UK comes from suckler herds and half from dairy herds (breeding phase: cull cows and calves for fattening). This means that beef from dairy-bred calves have 50% of the carbon footprint of suckler beef because all the emissions for a suckler cow are allocated to beef, whereas over 90% of emissions for a dairy cow are allocated to milk, approximately 5% to cull cow beef and 1% to beef calves (Magowan et al., 2020). This distinction between the breeding and finishing phase is important in the development of mitigation strategies, and to understand where extensification can be introduced. Intensive beef systems are more suitable for late-maturing bulls and steers, and aim to finish animals on cereals at 12 - 14 months of age, or silage at 14-16 months of age. Semi-intensive systems are suitable for all types of dairy-bred animals, and aim to finish animals at 18 months of age. Extensive systems have been found to be more suitable for early-maturing animals, and aim to finish animals at 24-30 months of age mainly on grass and grass silage (Magowan et al., 2020). Therefore, considering that the growth rate is negatively correlated to GHG emissions of the animals, the main differences between these management systems depend by the length of fattening period and the feed quality characterising the livestock farms (Williams et al., 2006).

We found that the conversion from temporary to permanent pasture can generate benefits related to soil health, animal wellbeing, recycling, and farm resilience synergies. Perennial ryegrass is a high-quality feedstuff, capable of withstanding intensive grazing in rotational grazing systems (Mccarthy et al., 2020). Knowledge gaps still exist, relating to the management and suitability of these swards in rotational grazing systems (Mccarthy et al., 2020). Limited research showed that the conversion from annual to permanent grassland may increase land requirement to maintain livestock productivity. In the UK this trade-off is unlikely to occur as more than 70% of the beef sector is already largely permanently pasture-based. This means that a common denominator is that the livelihood of livestock farms depends to the quality of these permanent grasslands (Schils et al., 2022), and on the public demand for their product.

Soil measures (Measure 1E)

In the CCC Balanced Pathway Scenario, soil measures include farm strategies, such as grass and legumes, cover crops, grass leys, that enhance the use of organic residues, and increased use of legume crops to fix soil N can reduce excess fertiliser use and emissions. From the agroecological point of view, the disappearance of diverse, semi-natural grasslands from agricultural landscapes is considered a major cause for decline in bird biodiversity in UK regions (Newton, 2004). In that respect, ley-arable, cover crops, and legume crops can generate the highest level of agroecological transition, addressing from 5 to 6 agroecological principles (Table 5). These AEFPs can generate win-win situations in arable farming that arise from the complementarity amongst the diverse elements that characterised these systems. Through intercropping is possible to increase productivity, stabilise seasonal outputs, reduce external inputs, and generate synergies that reduce the vulnerability of agroecosystems to biological disturbances such as pest and pathogens or climate disturbance such as floods and wind. However, these studies are based on correlative approaches, whereas the ecological processes involved are still not well-understood (Bretagnolle et al., 2011). The main issue with intercropping is that its success depends on the chosen arrangement and associated species (Malézieux et al., 2009). In addition, they can be difficult to manage due to competition for resources between the associated crops (Vandermeer et al., 1998).

Legumes have traditionally been intercropped with other crops, especially cereals for their ability to fix N biologically and increase SOM (Gaba et al., 2015). Although in this review we did not find direct benefits related to GHG mitigation in both legume crops and intercropping, a number of individual studies found that in certain circumstances the rotation legume-cereal can play an important role in reducing GHGs (Jensen et al., 2012; Jeuffroy et al., 2013). Therefore, legume crops intercropped with non-legume crops can stabilize yields (Cernay et al., 2018), and provide multiple ecosystem services such as biological control, soil C sequestration, pollination, and nutrient cycling (Gaba et al., 2015), enhancing the agroecological stage of intercropping. We found that ley-arable systems can generate several benefits introducing a high stage of agroecological transition in the farms (Table 5). However, the cessation of this AEFP can generate important trade-offs. The termination of grassland carried out with tillage practice can lead to losses of SOC stock and fluxes of N mineralization.

The implementation of perennial cereal and pasture cropping can potentially generate the same benefits that derive from conservation tillage practices, but addressing a broader level of agroecological principles (Table 6). Compared to annual crops perennial cereal have a positive effect on soil C stock (Ledo et al., 2020), due to less soil disturbance, and increase species richness. While pasture cropping allows a higher degree of control on weeds. It is important to note that the substitution of annual crops with perennial cereal may require the continuation of weed and pesticide operations (Table 5). The right balance between permanent crops and annual crops such as legumes could limit the proliferation of weeds reducing the need of chemical inputs. In contrast to annual grain crops, perennial cereals tend to have lower yields due to a lower harvest. Importantly, low growth rates may increase the risk for weed impacts and the need to increased weed control activities.

Among the soil measures outlined in the Balanced Pathway Scenario, the increase of SOM, the improvement of soil structure and the mitigation of soil erosion can be achieved by reducing soil disturbances and increasing organic inputs. In this context, meta-analysis studies show that the implementation of no-till and minimum tillage practice positively influence soil C stocks. Regarding organic inputs, we found that these can indirectly increase soil C storage by increasing net primary productivity (Ryals & Silver, 2013). These AEFPs, however, generate a relatively low level of agroecological transition at farm level when applied on their own (Table 6). Organic inputs, however, are difficult to optimise based on soil nutrient availability and crop demands, and depending by the agronomic conditions they can negatively impact CO_2 and N_2O emissions (Table 3). The long-term application of no-till and reduced tillage may have negative impacts on soil indicators such as GHG emissions, bulk density, soil compaction and soil erosion / run-off processes, which can have a negative impact on crop yields. This means that, depending on the agroecosystems and the climate conditions, these trade-offs may indirectly cause risks of environmental pollution in waters from potential soil nutrient losses. The combination of crop residue retention and reduced tillage could be used to reduce these trade-offs (Table 5).

7.2 Options to release agricultural land (Action 2)

To achieve net zero by 2050, the CCC recognises that deep emissions reduction in agriculture and land cannot be met without changes in the way UK land is used. The options considered in the Balanced Pathway Scenario shift land use from traditional agricultural production towards woodland creation and peatland restoration to reduce carbon and increase sequestration. Improving agricultural productivity both in cover crops and livestock is identified as one approach that would support this transition, and could reduce emissions by 1 Mt CO_2e in 2035 and 2050.

Higher agricultural productivity (Measure 2H)

Sustainable intensification in the livestock sector has been reported as the best strategy to spare land and halt biodiversity loss (Phalan et al. 2011). Enhancing animal productivity is usually a successful strategy for mitigating GHG emissions from livestock production systems. Converted per unit of animal product, higher-producing animals have lower GHG emissions than low-producing animals (Hristov, Ott, et al., 2013). Sustainable intensification, therefore, seems to be the single most effective GHG mitigating strategy, which may allow a reduction in animal numbers providing the same edible product output with a reduced environmental footprint. However, intensification may be regarded also as unsustainable in view of risks for the environment, such as soil degradation and losses of nutrients and pesticides, and to pose threats for biodiversity and human health (Smith et al., 2013; Smith et al., 2008).

Through measure 2H, the CCC suggest that increasing utilisation rates from the current 50% to 80% with paddock grazing can lead to a near-doubling in grass yields as measured by dry matter per hectare. This presents an opportunity to increase stocking rates without impacting feed requirements (quantity and quality) to enable some grassland to be used for other uses. From the agroecological point of view, we found that rotational grazing can cover up to 12 co-benefits spread across 6 agroecological principles. Depending on stocking rates, rotational grazing may generate high-low outcomes driven by increase in production and the environmental trade-offs due to soil compaction and nutrient losses (Table 5). In particular, it can improve soil C stocks, weight gain or milk yield per hectare. These effects are mostly driven by the approach characterising this AEFP, which allows the grazed paddock a rest period that permits forages to regrow, renews carbohydrate stores, improves yield and persistence that extends the grazing season.

The integration of agroecological principles has been reported to be easier in mixed farming systems as they already have the ability to recycle natural resources produced in the farms (Dumont & Bernués, 2014). In these agroecosystems, farmers already have the knowledge of both production systems, existing supply chains and routes to market. In mixed farms, the amendment of organic manure can contribute to the establishment of the recycling of nutrients produced within the farm boundaries In that respect, the flow of N within the farm boundaries is particularly relevant in agroecology, and the use of organic inputs needs to consider the centrality of natural resources produced on the farm. Internal sources include manure, crop residues, compost, litter and roots, as well as the soil N pool. External sinks or losses include N runoff and emissions of various compounds such as N_2O and ammonium, as well as the produce that leaves the farm. Importantly, N fixing by legumes in multispecies leys counts as an additional external source of N, as it produces reactive N from atmospheric molecular N. Grazing of grasslands returns the N captured by forage legumes directly back to the soil. Conversely, the production of hay and silage that remove N from the grasslands, are returned to the crop areas as manure at variable rates and times of the year. This means that, the grazing of multispecies leys enables farmers to naturally collect external N from rather low intensity systems. While, unless it is imported into the system, manure is not a source of N, but a means of storage and spatial redistribution of a natural resource produced in the farm. This separation between plant grazing / manure N can generate both advantages and disadvantages in the recycling efficiency of N (Lemaire et al., 2015), and needs to be clearly addressed in order to identify potential reductions in inputs and for optimising farm system metabolism.

Integrated crop-grassland-livestock systems allows for better management of nutrient flows and of landscape structures, with synergistic effects on biodiversity. Overall, multispecies leys system can generate several benefits introducing a high stage of agroecological transition in the farms. Multispecies leys have positive effects on livestock yields (beef and milk), and enhance soil C stocks. In addition, they have the potential co-benefit to reduce N losses from livestock, and support functional biodiversity within the farm enhancing soil fertility and water regulation.

In the Balanced pathway scenario, the measure “Higher agricultural productivity” targets also the enhancement of crop yields from an average of 8 t/ha currently to 11 t/ha. CCC advocated that this goal should be achieved adopting new technology and innovation, increasing farm water storage, improving soil fertility through agronomic practices, and increasing livestock utilization rates. A pre-requisite of this measure, however, is the avoidance of additional inputs such as fertiliser and pesticides, and increase farm water storage capacity. As described above, we think that AEFPs such as Intercropping, Rotational grazing, and Multispecies leys can generate win-win situations in measure 2H (Table 7)

7.3 Agroforestry and hedges (Action 4)

Silvoarable, silvopasture and hedges (Measure 4O, 4P)

The Balanced Pathways assumes that by 2050 between 5-15% of agricultural land can adopt silvoarable or silvopastoral systems, hedgerow length could be increased by between 30% to 40%, mostly in permanent and temporary grasslands to avoid soil C stocks changes. Here, we found that both silvoarable and vegetated strips can generate the highest agroecological transition of conventional agriculture (Table 5). In particular, silvoarable systems can increase agroecosystem C stocks (Table 3), and have been found to increase productivity compared with equivalent mono cropped land between 0.98 and 1.37 % (Graves et al., 2010). The complementary processes characterising these systems can generate several synergistic loops from the strengthening of biodiversity and healthier production systems improving biodiversity-dependent ecosystem services (e.g. creation of wildflower strips to promote pollination services). However, despite these benefits both AEFPs have a low level of integration in the UK. Silvoarable is the intercropping of trees or shrubs with arable crops, and timber is typically the main tree product produced in silvoarable systems, although intercropping with fruit trees could be achieved in temperate regions (Staton et al., 2019). Vegetated strips can be used to produce energy grasses, hay and silage, pasture for extensive grazing and biogas feedstock (Mayer et al., 2007), which can be relevant in the agroecological transition of mixed farms. The management of these systems however can be complex, and there is the risk that to make space for them land might be taken out of production. Silvoarable could increase the abundance of pests such as slugs relative to arable controls (Staton et al., 2019). Similar effects have been reported in and adjacent to flower-rich field margins (Eggenschwiler et al., 2013; Frank, 1998), suggesting that areas which provide a refuge from tillage could boost slug populations. Although research indicates that agroforestry systems can be agronomically viable, results are variable and further research is required (Albrecht et al., 2020); (Tscharntke et al., 2021).

8 Evidence gaps and directions for future work

The stability and resilience of the ecosystems is regulated by nine key processes at global and local scale, the boundaries of which should not be transgressed by human activities (Rockström et al, 2009). At global scale, however, intensive agriculture has already crossed the boundaries of five of these nine planetary boundaries. They are green/blue water use[1], land-use change, chemical pollution, biodiversity loss, and altered biogeochemical cycles (P and N). Freeing-up land for forestry, bioenergy crops and carbon farming through more efficient and productive systems alone might be not sufficient to achieve the required sustainability of agriculture in the transition to net zero, and its resilience to climate change. There is a growing realisation that both sustainable agroecosystems and food security is a matter of how food is produced in relation the local availability, access, utilization and stability of natural resources, inputs, market and services (HLPE, 2017b, FAO, 2018b). In this context, by improving efficiency in the use of natural resources as sustainably as possible, agroecology offers great potential to reduce the environmental impact of agriculture (De Schutter and Vanloqueren, 2011).

In this report, we reviewed the environmental impact of nineteen agricultural practices that can be classified as agroecological. In particular, we explored their impact on eight indicators: GHG emissions, terrestrial C stocks, productivity, recycling, input reduction, soil health, animal health, and biodiversity. We qualified these farm practices as more or less agroecological depending on the extent to which their implementation can benefit a wide range of environmental outcomes including biodiversity, carbon sequestration and storage, and GHG emissions. We found that agricultural practices such as legume crops in rotations, cover crops, perennial cereals, rotational grazing, multispecies leys, ley-arable rotations, intercropping, and vegetated strips can generate the highest level of agroecological transition at farm level (Table 5). The majority of these agroecological practices, however, have only a low or medium implementation level in current UK agriculture. This means that, through their implementation there is great potential to improve the sustainability of conventional agriculture. More research is required to understand the full impact of less-developed agroecological measures, including the potential for unintended consequences. Of the nineteen agroecological practices reviewed here, we found information on the impact of GHG emissions only on seven practices. In addition, for the majority of the practices the understanding of their impact at regional level on productivity and terrestrial C stocks is limited by the low level of confidence of the information found in global scale studies.

[1] Threats to human livelihoods due to deterioration of global water resources are threefold: (i) the loss of soil moisture resources (green water) due to land degradation and deforestation, threatening terrestrial biomass production and sequestration of carbon, (ii) use and shifts in runoff (blue water) volumes and patterns threatening human water supply and aquatic water needs, and (iii) impacts on climate regulation due to decline in moisture feedback of vapor flows (green water flows) affecting local and regional precipitation patterns (Rockström et al, 2009).

Assessing the environmental, economic and social impacts of agroecology in conventional agriculture, and how to scale out agroecological approaches must be an active area for research in the UK. Since 2010, in the UK the investment in research for agroecological projects represented less than 5 percent of agricultural research projects, and less than 0.5 percent of its total research budget since 2010 (Pimbert and Moeller, 2018). Targeting funding towards agroecological research would support generation of evidence to facilitate consideration of AEFPs in future emission reduction scenarios. Here, we found significant scientific knowledge gaps in understanding the relative impact of agroecological practices on several indicators when compared to other alternatives. There is also still limited applied knowledge and on-farm experience with practices such as intercropping, pasture cropping. While N can be biologically fixed by incorporating legumes in cropping practices, knowledge gaps exist on sustainable strategies for replacing the P which is removed with crop (HLPE, 2019). Perennial cereal crops are at the development stage, resulting in little data available from field experiments about how perennial cereals may adapt on marginal land (Audu et al., 2022). Little or no evidence was found of effects of agroforestry on pest control or pollination services (Staton et al., 2019). We lack of pair-wise studies on the transition from permanent to temporary grasslands (Schils et al., 2022). We have limited knowledge on the management and suitability of multispecies leys in rotational grazing systems. Long-term grazing studies based on multispecies swards are needed to understand the animal performance, sward persistence, and changes in plant nutritive value over time (McCarthy et al., 2020). In this context, we need to identify the differences within pasture-based production systems where different combinations of plant species and management strategies may influence meat and milk yields and their quality (Lee et al., 2021). This impact on yields, in particular, is an important gap in the literature given the great role that diversified pastures will likely play in net zero.

The knowledge gaps outlined in Table 3 and Table 5 represent an important limitation for system-based modelling research that explores multiple outcomes, and identifies where synergies and trade-offs could occur when developing agroecological strategies. The modelling of agroecological systems need to consider the complexity of diversified cropping and livestock systems and multifunctional land uses, in relation to historical knowledge and agroecological principles. Therefore, modelling these complex agroecosystems implies taking into account the network of interactions between plants and animals, pest and diseases, regulating communities, the flow of natural resources at system level (water, nutrients and radiative resources) and the resilience of these systems. In this context, future work that should receive the highest priority for funding are:

1) Modelling of low-input high-output agroecosystems beyond raw yield predictions and their environmental outcomes. Modelling agroecological farming should integrate site level information on diversified cropping systems and biotic stresses (such as the effect of pest and diseases and their regulations), which are key processes when dealing with zero pesticide systems.
2) Examining (by modelling) the impact of agroecological practices on food production so that any implications for agricultural output can be quantified and adequately considered.
3) To support targeting of agroecology in the farmed landscape, new experimental and modelling research is needed on multifunctional land uses in conventional agroecosystems to assess optimal plant/animal communities, and the spatio-temporal organization that optimizes its resilience to pest and diseases and to inter-annual variations. This could include the integration of process-based and statistical models to address the issue of species diversification or spatial heterogeneity.
4) Including the output from the above modelling research in biomass balanced methods to determine the impact of agroecology on the UK country-specific food balance, and the Balanced Net Zero Pathway Scenario of CCC.
5) More research is need to understand the socio-economic factors that support/limit the agroecological transition of conventional farming systems. The most important parameters for a limited or broader application of agroecology today are the farming knowledge and practical

experience about agroecological practices, and the economic cost that conventional farm need to sustain when system change or redesign of cropping systems are required.

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1 APPENDIX A

1.1 Literature search details

Searches were conducted in Web of Science between 2-7 March 2022, for 'all years' in the following databases:

Web of Science Core Collection (1900–present)

Science Citation Index Expanded (1900–present)

Social Sciences Citation Index (1900–present)

Arts & Humanities Citation Index (1975–present)

Conference Proceedings Citation Index-Science (1990–present)

Conference Proceedings Citation Index-Social Science & Humanities (1990–present)

Book Citation Index–Science (2005–present)

Book Citation Index–Social Sciences & Humanities (2005–present)

Emerging Sources Citation Index (2015–present)

Current Chemical Reactions (1986–present) (includes Institut National de la Propriete Industrielle structure data back to 1840)

Index Chemicus (1993–present)

BIOSIS Citation Index (1969–present)

Current Contents Connect (1998–present)

Data Citation Index (1993–present)

Derwent Innovations Index (1993–present)

KCI-Korean Journal Database (1980–present)

MEDLINE® (1950–present)

Russian Science Citation Index (2005–present)

SciELO Citation Index (2002–present)

Zoological Record (1993–present)

Only studies published in English were included, as these were most likely to be generalisable the context of interest (i.e. the UK)

We used the following search strings, structured by Outcome, Location, Study Type and Intervention (OR Boolean operators between each term within search string component, AND operators between components). All components were searched for using 'Topic' in WoS (title, abstract and keywords), apart from the Study Type terms which were restricted to 'Title' to reduce screening effort required. Terms in grey text were only included for interventions where these were relevant. A Location search string was included to prioritise results most generalisable to the UK context (i.e. based in European or temperate climates). A Study Type search string was used to prioritise meta-analyses and quantitative syntheses within the search results, but these terms were removed from the search where no meta-analyses could be identified in order to move down the evidence hierarchy. Interventions were searched for individually rather than simultaneously (i.e. separate search queries and screening for each intervention).

Outcomes

[Emissions] “carbon balance” OR “greenhouse gas*” OR emission* OR GHG OR “global warming potential” OR “GWP*” OR CO2 OR “carbon dioxide” OR N2O OR “nitrous oxide” OR CH4 OR methane

OR

[Carbon stocks] “soil organic carbon” OR “soil carbon” OR “soil C” OR “soil organic C” OR SOC OR “carbon pool” OR “carbon stock” OR “carbon stocks” OR “carbon storage” OR “soil organic matter” OR SOM OR “carbon sequestrat*” OR “C sequestrat*” OR “aboveground biomass”

OR

[Productivity] yield* OR harvest* OR return* OR perform* OR productivity OR production OR biomass OR herbage OR forage OR “dry matter” OR DM OR “DM/ha” OR growth OR liveweight OR “live weight”

Location

UK OR “United Kingdom” OR Britain OR British OR England OR English OR Scotland OR Scottish OR Wales OR Welsh OR Ireland OR Irish

OR

temperate OR Europe* OR France OR French OR Germany OR German OR Belgium OR Belgian OR Netherlands OR Holland OR Dutch OR Denmark OR Danish OR Spain OR Spanish

Study type

meta-analys* OR metaanalys* OR “meta analys*” OR “meta*analys*” OR “meta-regression*” OR “systematic review” OR “systematic* review*” OR synthes*

Intervention (searched separately in combination with search components above)

Zero or minimum tillage

till* OR “no till*” OR “reduced till*” OR “direct drill*” OR “conservation till*” OR “minimum till*”

Cover crops/Integration of arable cropping and livestock (e.g. grass in arable rotation)

rotat* OR “break crop*” OR grass* OR clover OR clovers OR ley* OR legum* OR “cover crop*” OR “grass clover” OR “crop* system*” OR fallow* OR “set*aside” OR “catch*crop*” OR intercrop* OR “green manur*” OR perennial* OR “mixed farm*” OR “crop-livestock” OR “fertility build*” OR “under*sow*” OR sheep OR ewe* OR lamb OR lambs OR "Ovis aries" OR ovine OR “Bos taurus” beef OR dairy OR cattle OR cow OR bull OR steer OR heifer OR cows OR bulls OR steers OR heifers OR calf OR calves OR bovine OR ruminant OR ruminants OR livestock

OR grazed OR grazing OR graze OR pasture OR pastures OR pastoral OR herbage OR sward* OR “sod based” OR hay OR silage OR forag* OR fodder OR cut OR cutting

Organic manures (replacing chemical fertilisers) compost OR slurry OR manur* OR muck* OR FYM OR residue* OR “organic fert*” OR “organic residu*” OR “organic manur*” OR “organic amend*” OR “organic wast*”

Retention of straw

(residu* OR straw OR crop OR mulch) AND (manag* OR residu* OR retention OR retain* OR remov* OR incorporat* OR return* OR applicat* OR apply*)

Use of minerals to maintain soil pH (e.g. lime)

lime OR liming [dropped Study type terms and used (analys* OR assess* OR estimat* OR review* OR synthes*) instead for this intervention]

Legume crops included in rotation

(arable OR agricult* OR farm* OR crop* OR cultivat* OR rotat*) AND (legum* OR pulse$ OR lucerne OR alfalfa$ OR lupin$ OR bean$ OR pea$ OR lentil$ OR clover OR soy OR soybean$)

Multiple cropping

Intercropping OR “pasture cropping”

Perennial crops (typically woody, not including grass in arable rotation)

perennial* OR “perennial crop*” OR biomass OR bioenergy OR “energy crop*” OR “woody crop*” OR Miscanthus OR switchgrass OR “short rotation coppice”

Field-edge buffer strips

"field edge*" OR "buffer strip*" OR "field margin*" OR "riparian strip*"

Silvoarable (intercropping woody vegetation within an arable system)/Silvopasture (woody vegetation within a livestock system)

agro$forest* OR silvo$pasture* OR silvo$arable* OR shelterbelt* OR “alley crop*”

Conversion of temporary leys to permanent pasture

(grass* OR pasture$ OR ley$ OR sward$) AND (permanent AND temporary)

Extensive grazing, i.e. reduced stocking density

“stock* densit*” OR “stock* intensit*” OR “stock* regime$” OR “graz* densit*” OR “graz* intensit*” OR “graz* regime$” OR “graz* manag*”

Rotational grazing practices/Multi-species swards

“rotat* stock*” OR “mob stock*” OR “mob-stock*” OR “cell* stock*” OR “control* stock*” OR “paddock* stock*” OR “holistic* stock*” OR “plan* stock*” OR “manag* stock*” OR “strip* stock*” OR “adaptive* stock*” OR “precision stock*” OR “regenerative stock*” OR “densit* stock*” OR “intens* stock*” OR “stock* intens*” OR “stock* densit*” OR “rotat* graz*” OR “mob-graz*” OR “mob graz*” OR “cell* graz*” OR “control* graz*” OR “paddock* graz*” OR “holistic* graz*” OR “plan* graz*” OR “manag* graz*” OR “strip* graz*” OR “adaptive* graz*” OR “precision graz*” OR “regenerative graz*” OR “densit* graz*” OR “intens* graz*” OR “graz* intens*” OR “graz* densit*” OR “multi*paddock” OR “fertility build*” OR multispecies OR “multi-species” OR “pasture composition” OR “sward composition” OR “species composition” OR “species diversity” OR “diverse mix*” OR “diverse sward*” OR “mixed sward*” OR “mixed ley*” OR “species mixture” OR “pasture species” OR “grassland species” OR “forage species” OR “deep* root*”

Reduced supplementary feed inputs/increase grass content of diet

(concentrate$ OR feed OR cereal* OR supplement*)

AND

(forage OR grass* OR herbage OR silage OR hay OR fodder OR pasture OR graz*)

AND

(ruminant OR ruminants OR livestock OR sheep OR ewe OR ewes OR lamb OR lambs OR "Ovis aries" OR ovine OR “Bos taurus” OR beef OR dairy OR cattle OR cow OR bull OR steer OR heifer OR cows OR bulls OR steers OR heifers OR calf OR calves OR bovine)

Citation title and abstract screening was conducted in Web of Science, and full texts for relevant citations were retrieved primarily via Google Scholar.

1.2 Data harmonisation

Quantitative estimates extracted from meta-analyses and other studies were harmonised to Response Ratios (RR):

RR = (XT/XC)

where

RR: response ratio

XT: treatment mean

XC: control meanA

Where meta-analyses presented estimates for treatment and control, these were converted to RR regardless of their original units. Where natural log (ln) RR were presented by studies, these were converted to RR relative to 1 by computing e raising to the power of lnRR (i.e. e^lnRR). Where meta-analysis estimates were presented as a proportional increase in the outcome relative to the control, these were converted to RR by assuming XC = 1, and XT = 1+(%effect/100).

Where estimates were not provided for control practices and intervention estimates were presented in raw units (e.g. t/ha), standard baseline/control estimates were applied to calculate response ratios. UK arable land was assumed to have a baseline soil carbon stock of 50 t/ha at 0-30 cm depth based on previous UK and European modelling work (Jordon et al. 2022, Smith et al. 2000). UK hedgerows were assumed to have an average height of 3m (Axe et al. 2017). Baseline UK grain yield was assumed to be 8 t/ha/year. To calculate baseline N_2O emissions, UK average fertiliser rates were used (137 kg N/ha for crops and 54 kg N/ha for grassland (Defra, 2018)). The IPCC Tier 1 default for direct emissions (N-N_2O) is 1% of applied N, and this was converted to N_2O using a conversion factor of 1.571429 (i.e. molar mass of N_2O relative to N-N_2O). This was then converted to carbon dioxide equivalents (CO_2e) using the GWP100 value 298. This gave baseline N_2O emissions of 0.642 t CO_2e/ha/yr for arable, and 0.253 t CO_2e/ha/yr for grassland. Where conversion between rate of change (i.e. per year) and absolute change was required, and estimate duration (e.g. of experiments in underlying studies) was not provided in articles, a 20-year time horizon was used as per IPCC Tier 1 default methodology.

Where available, estimate standard deviation (SD) or standard errors (SE) were harmonised to 95% Confidence Intervals (95% CI), using the below formulae:

SE = SD/sqrt(n)

95% CI = estimate mean ± (SE * 3.92/2)

Original units, harmonisation processes undertaken, and corresponding assumptions made, are all recorded in the data extraction sheets (Appendix B).

2 APPENDIX B

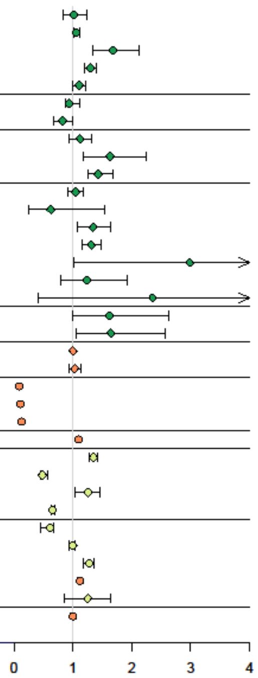
Dataset used in the calculations

https://docs.google.com/spreadsheets/d/1T3-QAMo8Pt176Q4NMyCgn5geAt4kXT3L/edit?usp=sharing&ouid=113687719835878263836&rtpof=true&sd=true

3 APPENDIX C

Figure B.18: Impact of agroecological farming practices (interventions) on greenhouse gas emissions. Greenhouse gases (GHGs) considered are carbon dioxide (CO_2), nitrous oxide (N_2O) and methane (CH_4). Results are presented and plotted as response ratios, calculated as the intervention mean divided by control mean, where a RR of 1 represents no effect of the intervention relative to control, RR < 1 represents a decrease in outcome following adoption of intervention (i.e. lower emissions), and RR > 1 represents an increase (i.e. higher emissions), along with 95% Confidence Intervals. Note that no practices created a net sink of any greenhouse gas(es). Where confidence intervals are not presented, this is because error parameters were not presented in the underlying study. The interventions of pasture cropping, perennial cereals, intercropping, legume crops, ley-arable rotations, silvopasture and rotational grazing are not presented here as no GHG data was identified.

Intervention	Citation	GHG	n	Response ratio	Confidence interval
No tillage	Sanden et al. (2018)	CO2	12 (53)	1.01	[0.83, 1.24]
	Shakoor et al. (2021)	CO2	9 (38)	1.06	[1.01, 1.11]
	Sanden et al. (2018)	N2O	16 (90)	1.68	[1.33, 2.13]
	Shakoor et al. (2021)	N2O	9 (38)	1.30	[1.20, 1.40]
	Shakoor et al. (2021)	CH4	9 (38)	1.10	[1.00, 1.22]
Minimum tillage	Sanden et al. (2018)	CO2	6 (34)	0.94	[0.88, 1.11]
	Sanden et al. (2018)	N2O	9 (53)	0.82	[0.68, 1.00]
Crop residue retention	Sanden et al. (2018)	CO2	3 (24)	1.12	[0.94, 1.33]
	Sanden et al. (2018)	N2O	4 (37)	1.63	[1.18, 2.25]
	Li et al. (2021)	N2O	(178)	1.43	[1.25, 1.68]
Organic manure: FYM	Sanden et al. (2018)	CO2	2 (4)	1.04	[0.92, 1.18]
	Sanden et al. (2018)	N2O	3 (9)	0.62	[0.25, 1.54]
	Zhou et al. (2017a)	N2O	(61)	1.34	[1.07, 1.65]
Organic manure: slurry	Sanden et al. (2018)	CO2	3 (5)	1.31	[1.16, 1.48]
	Sanden et al. (2018)	N2O	6 (9)	2.98	[1.02, 8.72]
Organic manure: compost	Sanden et al. (2018)	CO2	3 (8)	1.24	[0.80, 1.92]
	Sanden et al. (2018)	N2O	2 (5)	2.35	[0.40, 13.79]
Cover crop	Sanden et al. (2018)	CO2	2 (3)	1.62	[1.00, 2.62]
	Sanden et al. (2018)	N2O	3 (7)	1.65	[1.05, 2.58]
Silvoarable	Smith et al. (2008)	CH4		1.00	[1.00, 1.00]
	Smith et al. (2008)	N2O		1.03	[0.94, 1.14]
Vegetated strips: grass	Falloon et al. (2004)	N2O		0.08	
Vegetated strips: hedge	Falloon et al. (2004)	N2O		0.10	
Vegetated strips: trees	Falloon et al. (2004)	N2O		0.13	
Multispecies leys	Henderson et al. (2015)	N2O		1.10	
Extensive livestock grazing	Tang et al. (2019a)	CO2	(14/28)	1.35	[1.28, 1.42]
	Zhou et al. (2017b)	N2O	(7/5)	0.48	[0.45, 0.58]
	Tang et al. (2019a)	N2O	(8/16)	1.26	[1.04, 1.46]
	Tang et al. (2019a)	CH4	(15/29)	0.65	[0.63, 0.69]
Grass−based diets	Lynch (2019)	CO2	11	0.61	[0.46, 0.68]
	Lynch (2019)	N2O	11	1.00	[0.94, 1.04]
	Lynch (2019)	CH4	11	1.28	[1.17, 1.35]
	Smith et al. (2008)	CH4		1.12	
	Clark & Tilman (2017)	CO2e	7	1.25	[0.86, 1.64]
Permanent pasture	Smith et al. (2010)	N2O		1.00	

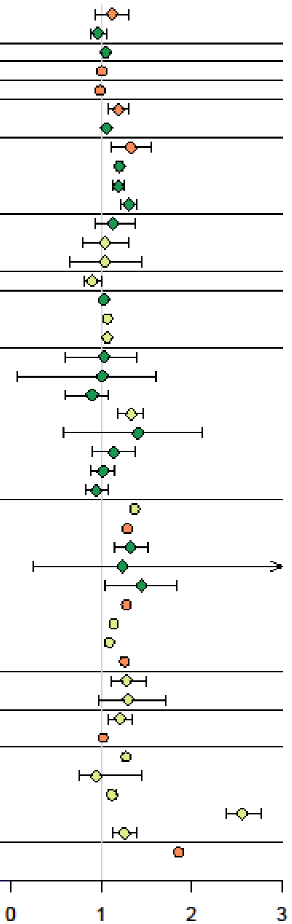


Footnote Figure B.1:

- *n corresponds to number of studies in meta-analysis estimates, with number of observations in brackets, where presented or applicable. For extensive livestock grazing, n corresponds to number of observations for light stocking density/number of observations for heavy stocking density.*
- *Dark blue points correspond to temperate meta-analyses (including Europe meta-analyses, temperate meta-analyses, and global analyses with temperate estimates), medium blue points indicated estimates from global meta-analyses without temperate-specific estimates, and light blue points represent modelling estimates (result from modelling simulation or parameter used as model input, typically generated from informal lit review) or primary studies (individual field study, typically non-UK).*

Figure B.19: Impact of agroecological farming practices (interventions) on soil organic carbon (SOC). Results are presented and plotted as response ratios, calculated as the intervention mean divided by control mean, where a RR of 1 represents no effect of the intervention relative to control, RR < 1 represents a decrease in outcome following adoption of intervention (i.e. lower SOC), and RR > 1 represents an increase (i.e. higher SOC), along with 95% Confidence Intervals. Where confidence intervals are not presented, this is because error parameters were not presented in the underlying study. The interventions of intercropping and grass-based livestock diets are not presented here as no SOC data was identified.

Intervention	Citation	Sampling depth (cm)	n	Response ratio	Confidence interval
No tillage	Powlson et al. (2012)	0–30	6	1.12	[0.94, 1.30]
	Sanden et al. (2018)		18 (76)	0.97	[0.88, 1.06]
Minimum tillage	Sanden et al. (2018)		32 (295)	1.05	[1.02, 1.08]
Pasture cropping	Badgery et al. (2014)	0–30	(24)	1.01	
Perennial cereals	Audu et al. (2022)	0–25	(3)	0.99	
Crop residue retention	Powlson et al. (2012)	0–30	4	1.20	[1.08, 1.31]
	Sanden et al. (2018)		41 (220)	1.06	[1.04, 1.08]
Organic manure: FYM	Powlson et al. (2012)	0–30	8	1.33	[1.11, 1.55]
	Sanden et al. (2018)		49 (194)	1.21	[1.17, 1.24]
Organic manure: slurry	Sanden et al. (2018)		32 (33)	1.19	[1.14, 1.25]
Organic manure: compost	Sanden et al. (2018)		11 (73)	1.30	[1.21, 1.40]
Cover crops	Sanden et al. (2018)		3 (7)	1.14	[0.94, 1.37]
Cover crops: legume	Abdalla et al. (2019)	0–30	(29)	1.05	[0.79, 1.31]
Cover crops: non-legume	Abdalla et al. (2019)	0–30	(13)	1.04	[0.65, 1.44]
Legume crops	West et al. (2002)	0–30	(12)	0.90	[0.81, 1.00]
Ley–arable	Jordon et al. (2021)	0–30	13 (70)	1.03	[1.02, 1.05]
	Conant et al. (2017)	0–20	3 (6)	1.07	
	West et al. (2002)	0–30	(18)	1.07	[1.04, 1.10]
Silvoarable	Ivezic et al. (2022)		24 (185)	1.03	[0.61, 1.40]
	Mayer et al. (2022)	0–20	9 (25)	1.01	[0.07, 1.60]
	Mayer et al. (2022)	20–40	9 (25)	0.90	[0.61, 1.08]
Agroforestry	Ma et al. (2020)	0–40	36 (98)	1.33	[1.18, 1.47]
Silvopasture	Mayer et al. (2022)	0–20	8 (10)	1.41	[0.59, 2.11]
	Mayer et al. (2022)	20–40	8 (10)	1.14	[0.90, 1.37]
	Chatterjee et al. (2018)	0–20	(7)	1.02	[0.89, 1.15]
	Chatterjee et al. (2018)	0–100	(15)	0.95	[0.83, 1.07]
Vegetated strips: grass	Van Vooren et al. (2017)	0–30	10 (108)	1.37	
	Falloon et al. (2004)	0–30		1.29	
Vegetated strips: hedge	Drexler et al. (2021)	0–28.4	10 (38)	1.32	[1.15, 1.51]
	Mayer et al. (2022)	0–20	3 (26)	1.24	[0.25, 3.10]
	Mayer et al. (2022)	20–40	3 (26)	1.45	[1.05, 1.84]
	Falloon et al. (2004)	0–30		1.28	
Arable 1.5m from hedge	Van Vooren et al. (2017)	0–27	10 (80)	1.14	
Arable 3m from hedge	Van Vooren et al. (2017)	0–27	10 (80)	1.09	
Vegetated strips: trees	Falloon et al. (2004)	0–30		1.26	
Rotational grazing (SOC)	Byrnes et al. (2018)		(44)	1.28	[1.11, 1.51]
Rotational grazing (AGC)	McDonald et al. (2019)		76	1.30	[0.98, 1.72]
Multispecies leys	Conant et al. (2017)	0–30	7 (19)	1.21	[1.08, 1.34]
	Henderson et al. (2015)			1.02	
Extensive livestock grazing (SOC)	Eze et al. (2018)	0–14.7	(100/65)	1.28	
	Abdalla et al. (2018)	0–30	27	0.94	[0.76, 1.45]
	Byrnes et al. (2018)		(56/59)	1.12	[1.08, 1.16]
Extensive livestock grazing (AGC)	Tang et al. (2019)			2.56	[2.39, 2.77]
	Zhan et al. (2020)		(26/29)	1.26	[1.13, 1.39]
Permanent pasture	Smith et al. (2010)	0–30		1.86	

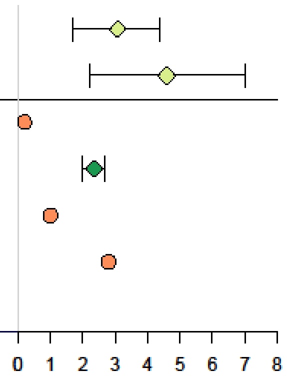


Footnote Figure B.2:

- *n corresponds to number of studies in meta-analysis estimates, with number of observations in brackets, where presented or applicable. For extensive livestock grazing, n corresponds to number of observations for light stocking density/number of observations for heavy stocking density.*
- *Dark blue points correspond to temperate meta-analyses (including Europe meta-analyses, temperate meta-analyses, and global analyses with temperate estimates), medium blue points indicated estimates from global meta-analyses without temperate-specific estimates, and light blue points represent modelling estimates (result from modelling simulation or parameter used as model input, typically generated from informal lit review) or primary studies (individual field study, typically non-UK).*
- *Organic manure: FYM stands for farmyard manure*
- *'Liming: low' refers to lime applications below 3 t/ha; 'Liming: medium' refers to lime applications in the range 3-5 t/ha; 'Liming: high' refers to applications above 5 t/ha*
- *For the intervention 'agroforestry', results include both silvoarable and silvopasture systems.*

Figure B.20: Impact of agroecological farming practices (interventions) on aboveground carbon (AGC) stocks. Results are presented and plotted as rate of carbon sequestration, in tonnes of carbon per hectare per year (t C/ha/yr), with 95% Confidence Intervals. Where confidence intervals are not presented, this is because error parameters were not presented in the underlying study. Only agroforestry and vegetated strips interventions are included as these are the only interventions considered in this report that have potential to substantially increase AGC stocks. Silvoarable is not presented here as no AGC data was identified.

Intervention	Citation	n	Sequestration (t C/ha/yr)	Confidence interval (95%)
Agroforestry	Ma et al. (2020)	23 (87)	3.08	[1.71, 4.40]
Silvopasture	Kim et al. (2016)	(10)	4.60	[2.20, 7.00]
Vegetated strips: grass	Falloon et al. (2004)		0.20	
Vegetated strips: hedge	Drexler et al. (2021)	5 (64)	2.35	[2.00, 2.70]
	Falloon et al. (2004)		1.00	
Vegetated strips: trees	Falloon et al. (2004)		2.80	

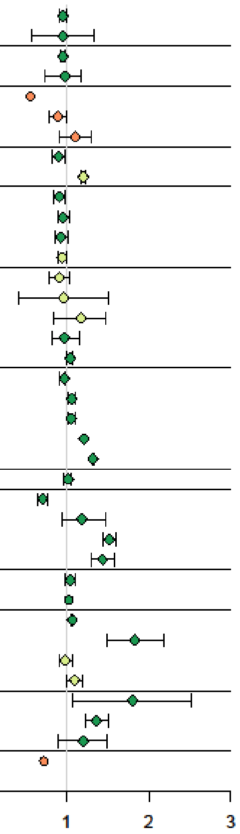


Footnote Figure B.3:

- *The sequestration rates presented by Falloon et al. (2004) are only applicable for grass strips for 1 year, and hedgerows for 5 years, after which time AGC is assumed to plateau.*
- *Aboveground carbon (AGC) results are presented for rotational grazing and extensive livestock grazing, as this is present in the herbage biomass rather than additional component to the system (plotted separately, Figure 3).*
- *For the intervention 'agroforestry', results include both silvoarable and silvopasture systems.*
- *n corresponds to number of studies in meta-analysis estimates, with number of observations in brackets, where presented or applicable.*
- *Dark blue points correspond to temperate meta-analyses (including Europe meta-analyses, temperate meta-analyses, and global analyses with temperate estimates), medium blue points indicated estimates from global meta-analyses without temperate-specific estimates, and light blue points represent modelling estimates (result from modelling simulation or parameter used as model input, typically generated from informal lit review) or primary studies (individual field study, typically non-UK).*

Figure B.21: Impact of agroecological farming practices (interventions) on agricultural productivity. Results are presented and plotted as response ratios, calculated as the intervention mean divided by control mean, where a RR of 1 represents no effect of the intervention relative to control, RR < 1 represents a decrease in outcome following adoption of intervention (i.e. lower productivity), and RR > 1 represents an increase (i.e. higher productivity), along with 95% Confidence Intervals. Where confidence intervals are not presented, this is because error parameters were not presented in the underlying study. The interventions of perennial cereals, legume crops, extensive livestock grazing and grass-based livestock diets are not presented here as no productivity data was identified.

Intervention	Citation	Productivity outcome	n	Response ratio	Confidence interval
No tillage	Sanden et al. (2018)	CY	19 (35)	0.95	[0.91, 1.00]
	Van den Putte et al. (2010)	CY		0.96	[0.59, 1.34]
Minimum tillage	Sanden et al. (2018)	CY	38 (97)	0.96	[0.93, 0.98]
	Van den Putte et al. (2010)	CY		0.98	[0.74, 1.17]
Pasture cropping	Millar & Badgery (2009)	CY	(3)	0.56	
Pasture cropping (high N)	Lawes et al. (2014)	CY	(3)	0.90	[0.79, 1.00]
Pasture cropping (low N)	Lawes et al. (2014)	CY	(3)	1.11	[0.91, 1.31]
Crop residue retention	Sanden et al. (2018)	CY	21 (35)	0.90	[0.82, 0.99]
	Liu et al. (2014)	CY	(66)	1.20	[1.17, 1.23]
Organic manure: FYM	Sanden et al. (2018)	CY	16 (60)	0.91	[0.85, 0.98]
Organic manure: slurry	Sanden et al. (2018)	CY	11 (37)	0.96	[0.89, 1.03]
Organic manure: compost	Sanden et al. (2018)	CY	6 (21)	0.93	[0.85, 1.02]
Organic manure	Shang et al. (2021)	CY	(9)	0.95	[0.89, 1.00]
Cover crop: legume	Abdalla et al. (2019)	CY	(52)	0.92	[0.79, 1.03]
Cover crop: non−legume	Abdalla et al. (2019)	CY	(96)	0.97	[0.42, 1.52]
Cover crop: mixed	Abdalla et al. (2019)	CY	(6)	1.18	[0.85, 1.48]
Cover crop	Sanden et al. (2018)	CY	6 (8)	0.98	[0.83, 1.16]
Catch crop	Sanden et al. (2018)	CY	24 (41)	1.04	[1.00, 1.08]
Undersown catch crop: legume	Valkama et al. 2015	CY		0.97	[0.92, 0.99]
Undersown catch crop: non−legume	Valkama et al. 2015	CY	7 (11)	1.06	[1.02, 1.11]
Undersown catch crop: mixed	Valkama et al. 2015	CY		1.06	[1.02, 1.11]
Intercropping: substitutive	Mahmoud et al. (2022)	CY	35 (167)	1.21	[1.21, 1.21]
Intercropping: additive	Mahmoud et al. (2022)	CY	35 (140)	1.32	[1.32, 1.33]
Ley−arable	Jordon et al. (2021)	CY	13 (70)	1.02	[0.97, 1.06]
Silvoarable	Ivezic et al. (2021)	CY	9 (67)	0.71	[0.64, 0.78]
Agroforestry	Torralba et al. (2016)		(19)	1.19	[0.95, 1.48]
Silvopasture	Pent (2020)	LER	19 (37)	1.52	[1.44, 1.60]
	Pent (2020)	LER	11 (25)	1.44	[1.30, 1.58]
Vegetated strips: pollinator mix	Lowe et al. (2021)	CY	7	1.04	[0.98, 1.10]
Arable 30m from hedge	Van Vooren et al. (2017)	CY	11 (343)	1.03	
Rotational grazing	Jordon et al. (2022)	HDM	9 (19)	1.07	[1.06, 1.07]
	Jordon et al. (2022)	LWG	8 (35)	1.83	[1.49, 2.18]
	McDonald et al. (2019)	LWG	79	0.99	[0.92, 1.07]
	McDonald et al. (2019)		36	1.09	[1.01, 1.19]
Multispecies ley	Jordon et al. (2022)	HDM	41 (174)	1.80	[1.08, 2.52]
	Jordon et al. (2022)	LWG	46 (184)	1.37	[1.23, 1.51]
	McCarthy et al. (2020)	MY	11 (25)	1.20	[0.90, 1.49]
Permanent pasture	Qi et al. (2018)	HDM		0.72	



Footnote Figure B.4:

- *Productivity outcomes are crop yield (CY), herbage dry matter (HDM), livestock liveweight gain (LWG), dairy cow milk yield (MY) and land equivalent ratio (LER, a metric for agroforestry systems that indicates the amount of land required to produce the equivalent agricultural and tree products if produced spatially separated rather than integrated). For Pent (2020), the first LER estimate corresponds to forage production plus the tree component in silvopasture systems, while the second estimate corresponds to animal production plus tree component. Blanks correspond to food production (Torralba et al., 2016) and animal yields per hectare (McDonald et al., 2019).*
- *n corresponds to number of studies in meta-analysis estimates, with number of observations in brackets, where presented or applicable. For undersown catch crops (Valkama et al., 2015), there were 7 studies (11 observations) across all categories (legume, non-legume and mixed).*
- *Dark blue points correspond to temperate meta-analyses (including Europe meta-analyses, temperate meta-analyses, and global analyses with temperate estimates), medium blue points indicated estimates from global meta-analyses without temperate-specific estimates, and light blue points represent modelling estimates (result from modelling simulation or parameter used as model input, typically generated from informal lit review) or primary studies (individual field study, typically non-UK).*
- *Organic manure: FYM stands for farmyard manure*
- *For the intervention 'agroforestry', results include both silvoarable and silvopasture systems.*