Evaluating agroecological farming practices

Report for Defra

Burgess. P.J., Redhead, J., Girkin, N., Deeks, L., Harris, J.A., Staley, J. (2023) Evaluating agroecological farming practices. Report from the “Evaluating the productivity, environmental sustainability and wider impacts of agroecological compared to conventional farming systems” project SCF0321 for DEFRA. 20 February 2023. Cranfield University and UK Centre for Ecology and Hydrology.

Photo on the front page is of an organic vegetable intercropping system with apple trees (Photo: Paul Burgess)

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List of acronyms

AC	Arable cropping
AES	Agri-environment Scheme
ASSET	ASSIST Scenario Exploration Tool
ASSIST	Achieving Sustainable Agricultural Systems
CC	Cover cropping
CO_2	Carbon dioxide
Defra	Department for Environment, Food and Rural Affairs
ERAMMP IMP	Environment and Rural Affairs Monitoring & Modelling Programme Integrated Modelling Platform
EU	European Union
EVAST	Environmental Value Assessment Scenario Tool
FABLE calculator	Food, Agriculture, Biodiversity, Land-Use, and Energy calculator
FAO	Food and Agriculture Organisation
GHG	Greenhouse gas
HLPE	High Level Panel of Experts of the Committee on Food Security
IFM	Integrated Farm Management
IPM	Integrated Pest Management
LEAF	Linking Environment and Farming
LCA	Life cycle assessment
LULC	Land use and land cover
N_2O	Nitrous oxide
NEVO	Natural Environment Valuation Online tool
PLFA	Pasture-fed Livestock Association
SFARMOD	Silsoe Whole Farm Model
SOC	Soil organic carbon
TAPE	Tool for Agroecology Performance Evaluation
UK	United Kingdom

Executive Summary

There are a range of definitions for agroecologically-related farming systems and practices. In brief, organic farming places strong restrictions on inputs, agroecological analyses often focus on principles, and regenerative farming typically emphasises the enhancement of soil health and the diversity of agricultural and wild species at a farm-scale. Perhaps surprisingly the role of agroecological systems in reducing net greenhouse gas emissions from food and farming is implicit rather than explicit. Despite some literature contrasting agroecological and technical approaches, many authors indicate that the desirability of farming practices should be determined by their impact at the appropriate scale. Sustainable intensification has been defined as maintaining or enhancing agricultural production while enhancing or maintaining the delivery of other ecosystem services. Approaches such as the Global Farm Metric and LEAF Marque Certification can support the integrated assessment of 12 groupings of attributes at a farm-scale covering inputs and outputs, and environmental and social impacts. In this report we reviewed the following 16 practices: crop rotations, conservation agriculture, cover crops, organic crop production, integrated pest management, the integration of livestock to crop systems, the integration of crops to livestock systems, field margin practices, pasture-fed livestock systems, multi-paddock grazing, organic livestock systems, tree crops, tree-intercropping, multistrata agroforestry and permaculture, silvopasture, and rewilding.

A review was completed of the measured effects of the above 16 agroecological practices or groups of practices on soil and biomass carbon, biodiversity, yields, costs, greenhouse gas emissions, and other impacts such as food value and labour use where available. From the literature reviewed, the report attempts as far as possible to determine the "mean" effect of a practice related to stated baseline, and the choice of baseline can be important. Obviously on individual farms, dependent in part on how the practice is implemented and the starting baseline, the individual response may vary from the mean. Moreover on an aggregated basis, the reviewed responses were assessed as being well-established, established but incomplete, unresolved, or inconclusive.

Most of the 16 agroecological practices led to increases in soil and/or biomass carbon and similar or enhanced levels of on-farm biodiversity (Table 7) relative to the stated baseline. However the effect on yields, the value of the product, input costs, and greenhouse gas emissions varies according to the specific practice and the baseline comparison. Therefore in most cases, a farmer implementing agroecological practices will need to balance trade-offs. In some cases, such as organic farming, a reduction in profitability due to a reduction in yield and certification costs may be compensated by an increase in product price. Where there are trade-offs, cost-benefit analysis is a potential tool to determine if the net effect at a farm-scale is positive from a societal perspective. Consequential life cycle assessment, which depends in large part on the yield per hectare, can be used to determine indirect off-farm effects, and the results depend in part on the assumptions made.

The uptake of agroecological practices at a farm-scale depends on the balance between the opportunities offered and the barriers to implementation. The opportunities created by agroecological practices, as described above, include increased soil carbon in surface layers, on-farm biodiversity, and increases in biomass carbon storage. The increasing requirements being placed on farm businesses by supermarkets and supply chains to develop "net zero food products" could be a durable and consistent driver for increased use of agroecological practices, but this needs to be

balanced by the strong drive for low food prices. The barriers to some agroecological practices will be geographical or incompatibility with management objectives at the farm-level. However, where these are not constraints, the major barriers are often related to uncertainty in the effect of the practices on yields and costs, and the need to finance the initial investment and certification costs. Enablers to overcome those barriers include knowledge exchange (particularly as the promotion of agroecological practices is not driven by organisations wanting to sell a product) and financial incentives (with a focus on market mechanisms that differentiate between desired and undesired societal outcomes).

There are existing frameworks that can be used to model the effect of wider uptake of agroecological practices at a UK scale such as ASSET, ERAMMP IMP, EVAST, and NEVO (See the main body of the text for details). However, this report identifies three barriers to their successful use. Firstly, modellers need to quantify the link between the considered scenarios and selected parameters within the underlying models. Secondly, as demonstrated in Section 3, the lack of readily available experimental data on the effect of agroecological practices and their change over time means that parameterising mechanistic or statistical models is challenging, and the alternative use of expert-based scoring or benefits transfer approaches can result in very large levels of uncertainty. Thirdly, an assessment of the aggregated impact of agroecological practices at a national scale will require an effective national monitoring approach that can assess the level of implementation of agroecological practices.

1 Introduction

Since 1960, crop and livestock production in the UK has primarily increased through specialisation (e.g. growing crops and livestock where they do best), improved genetic resources (e.g. crop varieties and livestock breeds), and reduction of crop and livestock stress (e.g. synthetic fertilisers, pesticides, improved livestock nutrition and housing, and veterinary care) (Burgess and Morris 2009). However specialised, intensive farming systems have also resulted in high regional concentrations of animals and manure, large-scale imports of feed; simplification of crop rotations; and high use of mineral fertilisers and pesticides. In turn this has been associated with high greenhouse gas (GHG) emissions, declines in soil quality (Graves et al. 2015), a 60% decline in the mean abundance of 214 “priority species” since 1970 (Hayhow et al. 2019), and the leaching and runoff of nutrients.

The UK Government has enacted legislation to only emit 22% of the 1990 value of net territorial greenhouse gas emissions by 2035, and to achieve net zero greenhouse gas emissions by 2050 (UK Government 2021). The Department for Environment, Food and Rural Affairs (Defra) has responsibility through the Environment Act (2018) to deliver a 25 year plan to improve the environment in England (HM Government 2018). The Climate Change Committee (2022) reports that “delivering on the Environment Act ambition, against the background of a changing climate, requires a coordinated approach across these targets and with other policy areas”. Defra is also responsible for delivering environmental land management schemes under the UK Agriculture Act (UK Government 2020a). In the recent UK Food Strategy (Defra 2022), there is a commitment to keep the current levels of domestic food production at “broadly the same level” at around 75% of what we consume.

Agroecological, and other low input and/or regenerative farming methods, have been proposed as a solution to enable reduced GHG emissions and agrochemical usage and improved soil health. However, the overall benefits can be difficult to establish. In a wider context, the European Union (EU) has released the Farm to Fork strategy which combines targets related to food consumption, climate change, biodiversity, fair economic returns in the food chain, and an increase in organic farming (European Union 2020). This includes an aim for 25% of total EU farmland to be under organic farming by 2030. The EU also has targets for greenhouse gas emissions associated with land use, land use change and forest (Simon 2022).

In the above context, the objectives of this study are:

1) To review definitions of agroecological farming, the metrics associated with sustainable agriculture, and identify UK-relevant agroecological farming practices.
2) To review the impacts of UK-relevant agroecological practices with a focus on soil health (primarily through their effect on soil carbon), on-farm biodiversity, food production, costs, and other ecosystem services including socio-economic and animal welfare impacts where available.
3) To review published evidence on the major opportunities for, barriers to, and enablers of agroecological innovations, technology and actions to improve productivity and sustainability.
4) To review and appraise the key tools to model agroecological vs non-agroecological systems including the use of spatial modelling and mapping and consideration of land-use availability and suitability.

2 Agroecological farming, metrics, and practices

2.1 Introduction

This section starts with a review of definitions of organic, agroecological and regenerative farming, and places them in the context of other terms such as sustainable intensification and climate-smart farming. It then examines the argument that the desirability or not of selected practices depends on their impact, which can be assessed using sustainability metrics. The final part of this section identifies 16 agroecological practices that have been proposed for use in the UK.

2.2 Organic farming

FAO and WHO (1999) define organic agriculture as "a production management system which promotes and enhances agroecosystem health, including biodiversity, biological cycles, and soil biological activity". However a key feature of organic agriculture is also the avoidance of synthetic fertilizers and pesticides (FAO and WHO 1999). In the UK, products can only be labelled as organic if at least 95% of the product's agricultural ingredients are organic, and all other ingredients and processing aids are permitted within the organic regulations (UK Government 2022c). The Organic Products Regulations (UK Government 2009) specifies that UK growers, processors and importers who sell feed and food as "organic" need to be registered with one of six approved organisations (UK Government 2020b). These regulations have strong rules on inputs. Whilst there are restrictions on some inputs, pesticides such as pyrethrin and copper sulphate are allowed under organic labels provided they are derived from natural rather than synthetic sources (Tscharntke et al. 2021). Likewise over-fertilisation can still occur with organic manures.

2.3 Agroecology

Application of the term "agroecology" varies between countries and contexts (FAO 2020), and hence it can be useful to be explicit in the definition being used (Wezel et al. 2009). For example "agroecology" can be defined in terms of science, as a social movement (HLPE 2019; Gliessman 2016, 2018, IPES Food 2022), and as a set of practices (Wezel et al. 2014).

The FAO (2018a) notes that "agroecology is an integrated approach that simultaneously applies ecological and social concepts and principles to the design and management of food and agricultural systems". Lampkin et al. (2015) in a review focused on the UK reported that 'agroecology' was the application of ecology to the management of agricultural systems at three levels of adoption: i) practices that emphasise functional biodiversity to reduce or replace external, synthetic, non-renewable inputs, ii) redesign focused on the farm ecosystem, and iii) a focus on agriculture as a human activity system. Similarly, the HLPE (2019) indicate that the degree to which an agricultural practice is agroecological depends on the extent to which: "(i) they rely on ecological processes as opposed to purchased inputs; (ii) they are equitable, environmentally friendly, locally adapted and controlled; and (iii) they adopt a systems approach embracing management of interactions among components, rather than focusing only on specific technologies".

The HLPE (2019) identified 13 agroecological principles, building on the FAO 10 elements, which in turn has similarities to the 10 principles described by the Landworkers Alliance (2019) (Table 1). Soil health, agricultural biodiversity, input reduction, and economic diversification are common technical and environmental features. Perhaps surprisingly, the FAO and HLPE definitions of agroecological systems do not make any specific mention of the role of the system in mitigating or adapting to climate change. By contrast, the Landworkers Alliance highlights climate change mitigation as an objective (Table 1).

Many of the definitions include a focus on the social and governance aspects of agroecological systems. For example HLPE (2019) stresses “the importance of local knowledge and participatory processes that develop knowledge and practice through experience, as well as scientific methods, and the need to address social inequalities”. Padel et al. (2017) following interviews with 14 farmers identified the importance of inspiration and social capital in agroecological systems. In a recent Scottish study, Lozada and Karley (2022) highlighted that agroecological farming is “more knowledge intensive and less reliant on chemical fixes” than conventional systems and there is usually a drive to use social mechanism to integrate farms more closely with local communities. According to HLPE (2019), this focus on governance issues has “profound implications” for how research, education and extension related to agroecological systems are organised.

Table 1. The 13 agroecological principles described by HLPE (Modified from HLPE 2019; page 41) categorised as environmental and technical or social and governance, and the relationship with the ten elements described by FAO, and 10 principles by the Landworkers Alliance (2019)

HLPE (2019) agroecological principles	FAO element	Landworkers Alliance (2019)
Environmental and technical		
Soil health: secure and enhance soil health for improved plant growth, by managing organic matter and soil biological activity.	Soil health	Building soil health
Biodiversity: maintain and enhance genetic, species, and functional diversity and overall agroecosystem biodiversity at range of scales.	Agricultural biodiversity	Encourage biodiversity
Input reduction: reduce or eliminate dependency on purchased inputs and increase self-sufficiency.	Exposure to pesticides	Replace agrochemicals
Economic diversification: diversify on-farm incomes thereby supporting greater financial independence for farmers.	Added value	Enhance economic resilience
		Climate change mitigation and adaption
Recycling: preferentially use local renewable resources and help close resource cycles of nutrients and biomass.		Promoting close loop systems
Animal health: ensure animal health and welfare.		
Synergy: enhance positive ecological interactions amongst the elements of agroecosystems (animals, crops, trees, soil and water).		
Social and governance		
Participation: encourage greater participation in decision-making and decentralised governance of agriculture and food systems.	Women empowerment	Integrating the community
Social values and diets: food systems based on the culture, social and gender equity of local communities that provide healthy, diversified, seasonally and culturally appropriate diets	Dietary diversity	Supporting culture and tradition
Fairness: support dignified and robust livelihoods for all actors based on fair trade, employment and intellectual property rights.	Income Productivity Youth employment	Affordability of food Quantity and quality of jobs
Land and natural resource governance: strengthen institutional arrangements to support of family farmers and smallholders.	Security of land tenure	
Co-creation of knowledge: including horizontal sharing of knowledge and farmer-to-farmer exchange.		Encourage innovation and education
Connectivity: ensure confidence between producers and consumers through fair and short distribution networks.		

2.4 Regenerative agriculture

In a review, Burgess et al. (2019) identified three main ways of defining regenerative agriculture including 1) a set of practices, 2) which may or may not avoid synthetic fertilizer and pesticides, and 3) a focus on going beyond the reduction of negative impacts to ensure that agriculture has a positive environmental effect.

2.4.1 Regenerative agriculture as a set of practices

The TED talk by Gabe Brown (2016) provides a good introduction to regenerative agriculture on his farm in northern USA, highlighting the importance of minimising cultivation and bare soil, encouraging diversity and water percolation, and integrating crop and livestock production at a farm-scale. Building on this, four common objectives that are widely associated with regenerative farming, and also highlighted by LaCanne and Lundgren (2018) and Cherry (2021), are: 1) abandoning tillage, 2) eliminating bare soil, 3) fostering plant diversity, and 4) integrating livestock and cropping operations. Additional objectives can include minimizing external inputs (see next section), keeping living roots in the soil (Cherry 2021), and encouraging water percolation into the soil (Savory and Duncan, 2016; Duncan 2016).

2.4.2 Regenerative organic agriculture

On his farm, Brown (2016) also highlighted no use of synthetic fertiliser and pesticides (Table 2). By contrast, Francis et al. (1986), Pearson (2007), and California State University (CSU 2017) report that regenerative agriculture seeks to minimize external inputs and negative external impacts outside the farm, and Lovins (2016) argued for a “circular economy of the soil”. In their analysis of methods to reduce GHG emissions, Drawdown (2017) recognised “regenerative agriculture” for annual cropping systems that include at least four of the following six practices: no-till or reduced tillage, cover crops, crop rotations, compost applications, green manures, and/or organic production (Table 2). Although their definition includes systems that are not “organic”, the associated technical notes imply that many systems are. To clearly differentiate between “regenerative agriculture” and organic production, the Rodale Institute used the term “regenerative organic agriculture” (Table 2).

Table 2. Some definitions of regenerative agriculture focus on the minimisation of fertilisers and pesticides, but some definitions (perhaps including the word organic) avoid their use

Practice	Brown (2016)	Regenerative agriculture		Regenerative organic agriculture
		CSU (2017)	Drawdown (2017)[a]	Rodale Institute (2018)
Minimise tillage	✓	✓	✓	✓
Minimise bare ground	✓	✓	✓	✓
Foster plant diversity	✓	✓	✓	✓
Increase water percolation	✓	✓		
Integrate crops and animals	✓	✓		Optional
Add green manures			✓	
Add compost			✓	
Avoid synthetic fertilizers and pesticides	✓	Minimise	Minimise	✓

Legend: ✓ means includes; a blank space indicates no data

[a]: Four of the six to be present

2.4.3 *Regenerative agriculture as farming that enhances*

Many current agricultural systems whilst providing safe nutritious food result in reduced soil fertility, carbon storage and biodiversity. Such systems could be termed “degenerative agriculture”. To address this, FAO (2014a) promotes “sustainable agriculture” that “conserves land, water, and plant and animal genetic resources, and is environmentally non-degrading, technically appropriate, economically viable, and socially acceptable” (Figure 1). An attraction of the term regenerative agriculture is that it provides an engaging narrative to promote change. In a similar way that a “circular” economy approach contrasts with a “linear” economy, regenerative agriculture can be contrasted with degenerative agricultural practices that degrade the soil and reduce biodiversity.

Figure 1. Regenerative agriculture aims to go beyond the “do no harm” principles of sustainable agriculture

Whilst some authors (e.g., Pretty et al. 2018) emphasise that sustainable agriculture also includes environmental enhancement, the specific focus of moving agriculture from being “non-degrading” to being “enhancing” is a particular focus of regenerative agriculture (e.g. Rhodes 2015). The Oxford English Dictionary defines regeneration as the “bringing of new and more vigorous life”. In the same way that many people want their life and their relationships to be more than “just sustainable”, many authors (Table 3) argue for a similar positive vision for agriculture. In the UK, the Food, Farming and Countryside Commission proposes “not just sustaining, but regenerating and restoring ecosystems” (RSA 2018). In some certification programmes, this regeneration extends beyond the environment to include enhanced human communities (General Mills, 2018).

Table 3. Definitions of regenerative agriculture focused on enhancement

Definitions of regenerative agriculture	Reference
• Farming and grazing practices that, among other benefits, reverse climate change by rebuilding soil organic matter and restoring degraded soil biodiversity – resulting in carbon drawdown and an improved water cycle.	California State University (2017)
• Regenerative agriculture actively builds the “system”, or resource base, it utilises.	Modified from Inwood (2012)
• A system of farming principles and practices that increases biodiversity, enriches soils, improves watersheds, and enhances ecosystem services.	Terra Genesis (2017)
• “Built on biological principles, regenerative agriculture seeks to concurrently enhance productivity and environmental management”.	Sherwood and Uphoff (2000)
• “For the system to be regenerative there must be an increase in both biodiversity and quantity of biomass”	Rhodes (2017)
• Agriculture that protects and intentionally enhances natural resources and farm communities.	General Mills (2018)

Building on the definitions in Table 3, Burgess et al. (2019) defined regenerative agriculture as "*a system of principles and practices that generates agricultural products, sequesters carbon, and enhances biodiversity at the farm scale*". The focus on soil health, carbon sequestration, and reversal of biodiversity loss were also the three main attributes identified in a study of the use of the term "regenerative agriculture" in the North of England (Magistrali et al. 2022).

2.5 Sustainable intensification and climate smart agriculture

Agroecological or nature-based farming practices are often contrasted with technology-based farming practices. For example, a UKRI and Defra supported MACSUR meeting at the Royal Society on 7 November 2022 posed the question as to whether "sustainable intensification" or "regenerative agriculture" offered the most promising pathway for agricultural sustainability? However, despite the title of the workshop most of the speakers indicated that this was a false binary choice. Instead they indicated that ideally the focus should be on the outcomes of specific practices rather than the process. For example, a related definition of sustainable intensification (that encompasses agroecological practices) may be the "maintaining or enhancing agricultural production while enhancing or maintaining the delivery of other ecosystem services". This focus on ecosystem services, rather than just environmental services allows the inclusion of societal aspects of sustainability (Diogo et al. 2022). Such an impact could be achieved by nature-led and/or technology-led sustainable intensification (SI) practices. This definition is also interesting in that, in a similar way to pareto-efficiency analysis, enhancing production whilst maintaining environmental services or enhancing environmental value whilst maintaining production is only possible if the current system is not pareto-efficient or there is a new technology or allocation of resources that allows the creation of a new food-environmental value boundary (Figure 2).

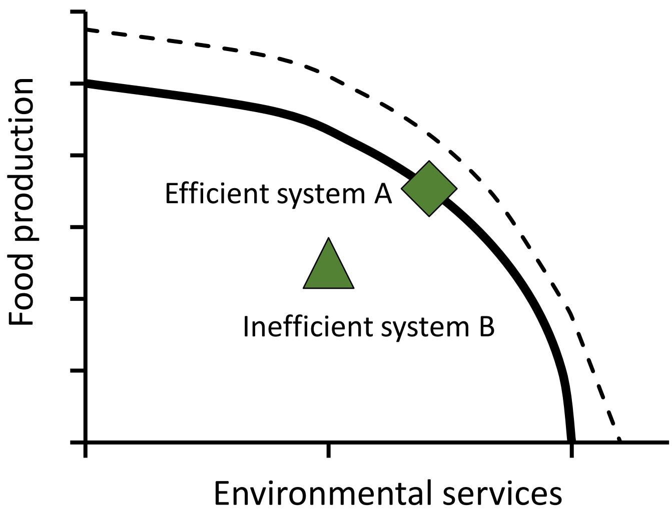

Figure 2. If current farming systems are pareto-efficient in terms of food production and environmental services (A), then it is not possible to increase food production without reducing environmental services or to enhance the environment without reducing food production. Hence sustainable intensification is only possible if the current system is not pareto-efficient (B) or new innovations or resources are introduced which allows the expansion of production-environment curve (dotted line).

In practice, innovations that can both increase yields and improve environmental impacts are less common than practices that increase yield but have negative environmental effects, or practices that have improved environmental effects but reduce yield. The use of cost-benefit analysis is one approach that allows decision makers to determine if the increase in say environmental health is sufficient to compensate for the reduction in yield. To undertake such cost-benefit analyses it is necessary to place an economic value on the added value of the production outputs and the environmental services (Burgess and Rosati 2018). For example, agroecological practices that increase the value of the cultural and regulating services of pig production by more than the decrease in provisioning services should be welcomed from a societal perspective (Figure 3).

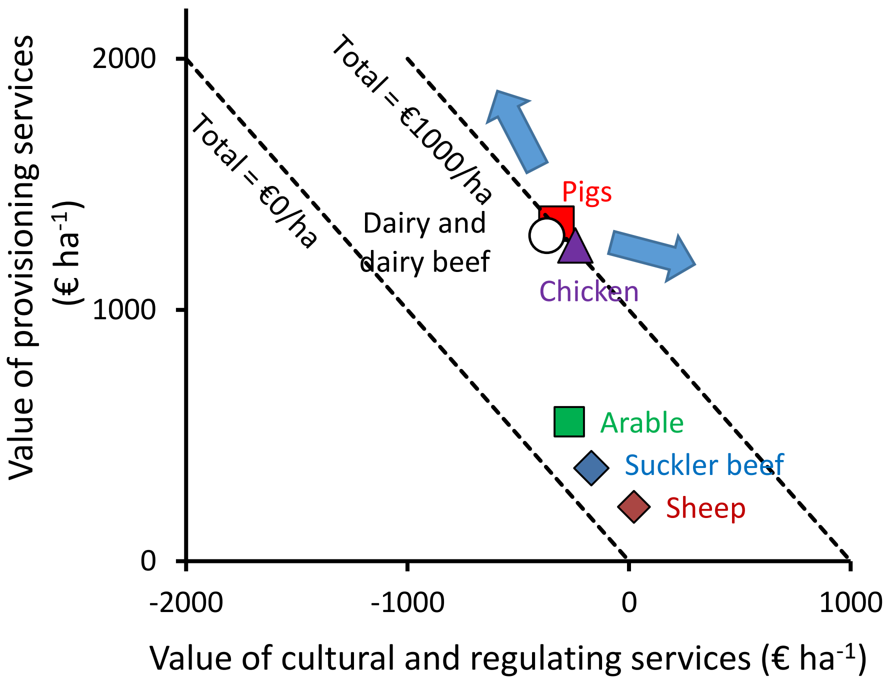

Figure 3. Annual value of provisioning services (y-axis) and cultural and regulating services (x-axis) of six UK farm systems (after Chatterton et al. 2015; assumption of 1.14€ = 1.00£). The combination of services creating similar combined values can be viewed as diagonal lines. The combined value of provisioning, cultural and regulating services for, for example pig production, can be increased by increasing the value of cultural and regulating services by more than the value of the reduction of provisioning services or vice versa (after Burgess and Rosati 2018)

An additional complexity in analysing the effect of agroecological practices on the overall value of provisioning, cultural and regulating services is the consequential effect of practices beyond the farm-gate. Weidema et al. (2018) argues that such effects should be considered. For example, the effects of a decline in UK production due to an agroecological practice can be assessed using consequential life cycle analysis, which assesses the consequential effects of market dynamics (Zamagni et al. 2012). Hence, Smith et al. (2019) predicted that a move to organic agriculture in England and Wales would result in a 6% reduction in agricultural GHG emissions in England and Wales, but the consequential effects of increased food imports, and increased conversion of grassland to cropland outside England and Wales, could lead to 0-56% greater GHG emissions when considered at a global level. These results assumed no changes in the UK diet and that reductions in UK production would be directly substituted by increased imports. Some authors have questioned these assumptions (van der Werf et al. 2020), and in practice, UK diets may change and interim high prices may reduce some aspects of food consumption.

One observation, as previously mentioned in relation to Table 1, is that the focus on agroecology does not specifically address the global climate emergency. It could be argued that there should be a greater focus on “climate-smart agriculture”. Climate-smart agriculture has been defined as “agriculture that sustainably increases productivity, resilience (adaptation), reduces/removes greenhouse gases (mitigation), and enhances the achievement of national food security and development goals” (FAO 2013). Hence in a similar way to sustainable intensification, there is an emphasis on increased production, but a major emphasis is also placed on reducing greenhouse gas emissions (Lipper et al. 2014). A recent review of the impact on agroecological practices is provided by Albanito et al. (2022).

2.6 A focus on impacts

Although there is a sustainable literature focused on definitions, many authors seem to emphasise that the important question is the extent to which these farming approaches can maintain or enhance food production whilst maintaining or enhancing environmental value, and some consideration of social impacts. The environmental values of agroecological systems typically places an emphasis on reducing greenhouse gas emissions, enhancing biodiversity, or improving soil health.

The Global Farm Metric (2022a) seeks to provide a common language and framework for the assessment of the sustainability of agricultural systems, and thereby inform practice and policy (Sustainable Food Trust 2022). At the end of 2022, it was updated to include 12, rather than 11 major categories (Global Farm Metric 2022b). The 12 categories can be grouped into four groups: i) inputs such as farmers and workers, nutrients, and resources, ii) outputs such as crops and pasture, animals, production, and economics, iii) environmental impacts such as nature, soil, water, and climate, and iv) community impacts (Figure 4). Most of these categories, with the possible exception of economics are also covered by Linking Environment And Farming (LEAF) Marque certification (Table 4). The Soil Association’s organic certification covers each topic except economics and community issues. The Red Tractor mark focuses on health and safety issues, and Pasture for Life and RSPCA Assured primarily focus on animal husbandry issues.

Figure 4. The Global Farm Metric comprises of 12 segments (Global Farm Metric 2022b)

Table 4. The 12 categories covered by the "Global Farm Metric" (Global Farm Metric 2022b) and the extent to which the components are covered by other sustainability metrics (after Sustainable Food Trust 2021), and TAPE (FAO 2019). Items indicated with a "✓" are included in the metric.

Global Farm Metric	Global Farm Metric	LEAF	TAPE	Soil Association	Red Tractor	Pasture for Life	RSPCA Assured
Farmers and workers	✓	✓	✓		✓		✓
Nutrients	✓	✓		✓	✓		
Resources	✓	✓		✓			
Crops and pasture	✓	✓		✓	✓		
Animals	✓	✓		✓	✓	✓	✓
Production	✓	✓	✓	✓	✓	✓	
Economics	✓		✓				
Nature	✓	✓	✓	✓		✓	
Soil	✓	✓	✓	✓	✓	✓	
Water	✓	✓		✓	✓		
Climate	✓	✓		✓			
Community	✓	✓	✓				

The Tool for Agroecology Performance Evaluation (TAPE) developed by FAO (2020) covers many of the same categories as the Global Farm Metric. Security of land tenure and income (which are included in TAPE) are also included as subcategories within the latest version of the Global Farm Metric (2022b) (Table 5).

Table 5. Ten core criteria in the Tool for Agroecology Performance Evaluation across five dimensions as described by FAO (2019) and Mottet et al. (2020)

Dimension		Core criteria of performance	Proposed method of assessment
Governance	1	Security of land tenure	
Economy	2	Productivity	£/ha and £/person
	3	Income	£
	4	Added value	£
Health and nutrition	5	Exposure to pesticides	
	6	Dietary diversity	
Society and culture	7	Women's empowerment	Women's empowerment in agriculture index
	8	Youth employment opportunity	Access to jobs, training, education and migration
Environment	9	Agricultural biodiversity	
	10	Soil health	SOCLA soil health method (Nicholls et al. 2004)

2.7 Selected agroecological practices

There are a wide range of agroecological practices. Serle (2017) studied the regenerative capacity of conservation tillage, cover cropping, enhanced crop rotations, residue retention, pasture cropping, and planned grazing. The Ellen MacArthur Foundation and SYSTEMIQ (2017) considered regenerative practices to include permaculture, organic agriculture, no-till polyculture, holistic grazing and keyline land preparation. In a study of 56 respondents in the North of England (Magistrali et al. 2022), regenerative practices were associated with crop diversification, cover crops, no- or minimum tillage, integration of livestock, integrated pest management, pasture-based livestock, agroforestry, organic practices, and the use of biosimulants. Building on these, practices described by Toensmeier (2016) and Drawdown (2017), we identified 16 agroecological practices (Table 6). Each practice meets at least two of the four objectives of minimising tillage, minimising bare soil, fostering plant diversity, integrating crops and animals, and a fifth objective of reducing synthetic fertilizers and pesticides (Table 6).

Table 6. Sixteen selected agroecological practices and how they include five regenerative agriculture objectives

System	Minimise tillage	Minimise bare soil	Foster plant diversity	Integrate crops and animals	Reduce synthetic fertilizers/ pesticides
Crop rotations			✓		✓
Conservation agriculture	✓	✓			
Cover crops		✓	✓		
Organic crop production	✓	✓	✓		×
Integrated pest management			✓		✓
Integrate livestock with crops	✓	✓	✓	✓	✓
Integrate crops with livestock			✓	✓	
Field margin management	✓	✓	✓		✓
Pasture-fed livestock systems	✓	✓	✓		✓
Multi-paddock grazing	✓	✓	✓	✓	
Organic grassland systems	✓	✓	✓	✓	×
Tree crops	✓	✓			
Tree intercropping	✓	✓	✓		
Multistrata agroforestry	✓	✓	✓		
Silvopasture	✓	✓	✓	✓	
Rewilding	✓	✓			

Legend: ✓ means necessary; × means prohibited; blank space means optional

Crop rotations: the frequent growing of the same annual crop on the same land often tends to result in yield decline (Bennett et al. 2012). This could be due to build-up of pests, diseases, and weeds, or nutrient depletion. One approach to address this problem is to rotate the growing of crops.

Conservation agriculture*:* is a cropping system with minimum tillage that ensures retention of crop residue mulch on the soil surface. Some definitions also include the diversification of plant species (Kassam et al. 2019) through intercropping, cover cropping, green manuring, and agroforestry, the integration of manure and organic materials, and judicious use of chemical fertilizers (e.g. Lal 2009).

Cover crops: are crops that are grown instead of maintaining a bare fallow during winter and the crop is typically ploughed in as a green manure before growing the next main crop (Poeplau and Don 2015). They are also known as “inter-crops” or “catch-crops”.

Organic crop production: the Rodale Institute (2018) uses the term regenerative organic agriculture to describe conservation agriculture that prohibits the use of pesticides and synthetic fertilizers. Whilst regenerative organic agriculture can include animals, it is not a specific requirement. Increased plant diversity is generally a feature of organic systems. Soil health, animal welfare and social fairness are specifically presented as three pillars of regenerative organic agriculture.

Integrated pest management: has been defined as the “careful consideration of all available plant protection methods and subsequent integration of appropriate measures that discourage the development of populations of harmful organisms and keep the use of plant protection products and other forms of intervention to levels that are economically and ecologically justified and reduce or minimise risks to human health and the environment” (European Union 2009; Barzman et al. 2015).

Integration of livestock in cropping systems: integrating livestock into arable systems can reduce dependence on external inputs of mineral fertilizers (Peyraud et al. 2014).

Integration of crops into livestock systems: conversely integrating crops into grassland systems can reduce dependence on external inputs of feeds and reduce nutrient losses (Peyraud et al. 2014).

Field margin practices include conservation headlands (where agrochemical use is reduced), field margins, hedgerows, set-aside, and wildflower strips.

Pasture-fed livestock systems: in the UK, the Pasture-Fed Livestock Association (PFLA) is an organisation that is encouraging “the raising of ruminant animals wholly on fresh or conserved pasture and forage” (Pasture-Fed Livestock Association 2017).

Multi-paddock grazing refers to rangeland management where the grazing unit has livestock on it for less than 10% of the time (Rhodes 2017). It is also known as “holistic planned grazing” (Teague et al. 2016) and has been called a regenerative practice (Lovins 2016; Teague and Barnes 2017). Like most grazing systems it minimises soil tillage and bare ground, but it also includes more complex rotations. It has also been termed “pulse grazing” and a “permaculture approach to rangeland management” (Rhodes 2017).

Organic grazing refers to certified organic livestock systems that prohibit the use of synthetic pesticides and fertilisers.

Woody perennial crops in the UK include horticultural crops like apples, pears and plums, which are anticipated to provide a higher store of carbon than arable and grass crops.

Tree intercropping, or silvoarable agroforestry, is the integration of woody perennials with arable or horticultural crops at field scale. The presence of trees reduces the need to cultivate the soil and plant diversity is typically increased.

Multistrata agroforestry is a farming system that integrates different layers of multiple woody perennials often with understorey herbaceous crops. It differs from multistrata forestry as food is an output. The presence of trees means that tillage and bare ground is minimised and plant diversity is increased. **Permaculture,** which was coined in the 1970s, is "an integrated, evolving system of perennial or self-perpetuating plants and animal species useful to man" (Mollison and Holmgren, 1981). Holmgren (2002) has also defined permaculture as "consciously designed landscapes which mimic the patterns and relationships found in nature, while yielding an abundance of food, fibre and energy". Whitefield (2011) reports that the inspiration for permaculture is to combine the self-reliance of a wood with the highly edible nature of a wheat field.

Silvopasture is the practice of integrating trees and the grazing of animals in a mutually beneficial way (Rodale Institute 2018). Because grass is largely a perennial crop, tillage and bare soil is minimised, and plant diversity is greater than conventional grassland.

Rewilding and agricultural land abandonment can mean different things in different locations. In America rewilding generally relates to the restoration of large wilderness areas with a focus on a dominant carnivore such as wolves (Corlett 2016). In this report, we use "rewilding" in the European sense of assisting the "regeneration of natural habitats through passive management approaches" (Navarro and Pereira 2015), which has also been termed "ecological rewilding". Rewilding is likely to minimise the extent of bare soil and it can include food production (Lorimer et al. 2015). The process may provide some opportunities for high value meat products and tourism.

2.8 Summary

There are a range of definitions for different farming systems and practices. In brief, organic farming places strong restrictions on inputs, agroecological analyses consider a range of principles, and regenerative farming places a heavy emphasis on soil health and biodiversity. Interestingly the definitions do not have a strong focus in relation to net zero targets. Despite some literature wanting to contrast agroecological and technical approaches, other authors indicate that the focus should be on the outputs. Sustainable intensification has been defined as "maintaining or enhancing agricultural provisioning while enhancing or maintaining the delivery of ecosystem services". Whilst some practices can increase yields and improve the environment, in practice many practices that improve the environment will result in a yield penalty. In such cases, a cost-benefit analysis can be used to determine if the net effect at a farm-scale is positive from a societal perspective. Consequential life cycle assessment can be used to determine indirect off-farm effects, and the results obtained depend in part on the assumptions made. Approaches such as the Global Farm Metric and LEAF Marque Certification consider groupings of attributes covering productivity, processes, environmental outputs, and social impacts at a farm-scale. Sixteen farming practices associated with agroecology are conservation agriculture, cover crops, organic crop production, crop rotations, integrated pest management, integration of livestock into arable systems, integration of crops into livestock systems, field-margin management, pasture-fed livestock, multi-paddock grazing, tree crops, tree-intercropping, multistrata agroforestry, silvopasture, and rewilding. The impact of each practice at a farm-scale is considered in Section 3.

3 Impacts of agroecological farming practices

3.1 Introduction

The eventual success of agroecological farming practices does not rest on their promise, but on their capacity to deliver on the ground. Some people are sceptical. For example, McGuire (2018) has defined regenerative agriculture as "conservation agriculture and holistic grazing plus exaggerated claims". This section reviews the impacts of UK-relevant agroecological practices with a primary focus on soil carbon, biomass carbon, biodiversity, yield, product value, costs, and greenhouse gases, with other health and welfare impacts mentioned where data are available.

3.2 Method

For each agroecological practice we built on a spreadsheet of evidence (Appendix A) based on the literature review reported by Burgess et al. (2019). Crop rotations, cover crops, integrated pest management, integration of crops and livestock, field margin practices, and pasture-fed livestock were added as new practices. When reviewing the practices, focus was placed on their impact in terms of quantifiable impacts on soil carbon (alongside biomass carbon and greenhouse gas emissions), biodiversity, and food production because of their direct link to government targets in relation to net zero greenhouse gas emissions, biodiversity, and proportional food imports. In general, the review did not focus on social and animal health impacts which are often assessed in qualitative rather than quantitative terms. The number of references was greatest for conservation agriculture (n = 21) and organic agriculture (n = 33) and least for rotations and tree crops (n = 6).

The level of confidence of impacts was based on the IPBES "four-box" model for qualitative communication of evidence (IPBES 2017, 2018), with the definitions being:

Inconclusive: existing as or based on a suggestion or speculation; no or limited evidence.

Unresolved: multiple independent studies exist but conclusions do not agree.

Established but incomplete: general agreement although only a limited number of studies exist but no comprehensive synthesis and, or the studies that exist imprecisely address the question.

Well established: comprehensive meta-analysis or synthesis or multiple independent studies that agree.

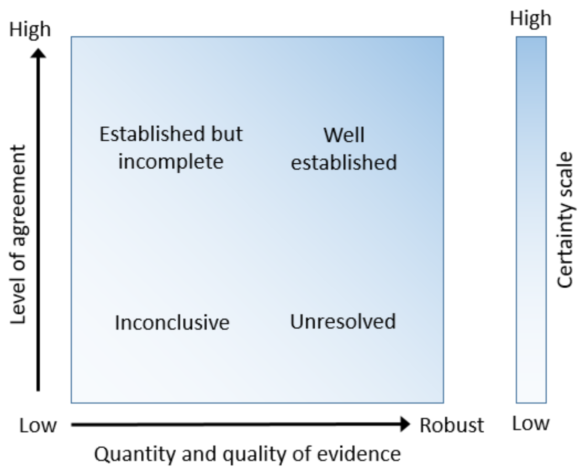

Figure 5. Four box model of the level of agreement and the quantity and quality of evidence (IPBES, 2018).

An important part of the method was to define a specific base-line or counterfactual for each intervention. For example organic agriculture may only provide mean yields of 0.68-0.90 of a well-fertilised and well-managed non-organic system (Lesur-Dumoulin et al. 2017). However, it can provide a yield equivalent to 1.43 to 1.87 of a non-fertilised control plot of sorghum in Africa (Tonitto and Ricker-Gilbert 2016). It is also important to note that the analyses focus on the **mean** response. For example, Lesur-Dumoulin et al. (2017) in a global meta-analysis also reported that whilst the mean yields of organic horticultural crops were 0.68 to 0.90 of non-organic crops, there was variation: with 10% of incidence resulting in only 50% of the yield, and a 20% chance of higher yields.

3.3 Results

The general effect of the practices, relative to a stated control, on soil carbon, biodiversity and yield are illustrated in Table 7. For each practice, a reference and value, subjectively selected by the authors as being representative after reviewing a selection of papers, is included. For example the value of 1.06 for the impact of crop rotations, relative to continuous cereal crops, implies that the soil carbon following crop rotations was 6% higher than with a continuous cereal crop.

Each of the 16 agroecological practices or group of practices generally lead to increases in soil carbon and similar or enhanced levels of on-farm biodiversity. However their effect on yields, input costs, and tree carbon and products varies according to the specific system and the baseline comparison. In general the analyses does not explicitly state the time period for responses to occur. For example a wild flower strip could be created in months, but substantial effects of tree planting on biomass carbon may take 10 years. Some response may show an ongoing effect, and some effects may eventually reach a plateau. Each practice is considered in turn.

Table 7a. Indicative main "on-farm" effects of 16 agroecological practices (expressed as effect of intervention divided by baseline with illustrative references). The colour of shading refers to whether the effect is positive, similar to positive, similar or very variable, similar to negative, or negative.

Agroecological Practice	Counterfactual or baseline	Soil carbon	Biomass carbon	On-farm biodiversity	Mean crop, grass or livestock yield	Input costs
Crop rotations	Continuous cereal cropping	1.06 (Liu et al. 2022)		1.03-1.15 (Venter et al. 2016)	1.05-1.37 (Angus et al. 2015)	Inconclusive
Conservation agriculture	Crop production with intensive tillage	Variable (Cai et al. 2022)		~1.00 (Doran 1980)	0.86-1.01 (Pittelkow et al. 2015)	Lower (Huggins and Reganold 2008)
Cover crops	Bare fallow	1.07-1.19 (Jian et al. 2020)	Higher	1.38 (Guzmán et al. 2019)	0.96-1.13 (Abdalla et al. 2019)	Higher (AHDB 2020)
Organic crop production with organic production with organic amendments	Crop production with fertilizers and/or agrochemicals	1.07-1.09 (Mondelaers et al. 2009; Tuomisto et al. 2012)		1.30-1.50 (Bengtsson et al. 2005)	0.48-0.92 (Clark & Tilman 2017; Cooper et al. 2016)	Lower to higher (LaCanne and Lundgren 2018; Crowder and Reganold 2015)
	Crop production with no amendments or fertilizers	1.07-1.09 (Mondelaers et al. 2009; Tuomisto et al. 2012)		Inconclusive	1.01-1.07 (Hijbeek et al. 2017)	Higher (Crowder and Reganold 2015)
Integrated pest management	"Baseline" pest management practice	Inconclusive		Higher or similar (Pecenka et al. 2021)	Higher or similar (Norton and Mullen 1994)	Reduced agrochemical costs
Integrated livestock/arable	Specialist arable	Similar (Cooledge et al. 2022)		Higher or similar (Tamburini et al. 2022)	0.93-1.02 (Peterson et al. 2020)	Inconclusive
Integrated livestock/arable	Specialist livestock	Decrease (Powlson et al. 2011)		Higher or similar (White et al. 2019)	Higher or similar (Dove et al. 2015)	Inconclusive
Field margin practices e.g. wild flower strips or hedges	Crop production	1.32 (Drexler et al. 2021)	[a]	2.7-7.1 (Batáry and Tscharntke 2022)	0.85-0.95[b] (Batáry and Tscharntke 2022)	Higher
	Grass production	0.91 (Drexler et al. 2021)	[a]	Variable (Kovács-Hostyánszki et al. 2011)	Inconclusive	Inconclusive
Pasture-fed livestock system	Grain-fed livestock system	Higher or similar		Higher or similar (Norton et al. 2022)	Lower (Herron et al. 2021)	Lower (Dillon et al. 2008)
Multi-paddock Grassland	Grassland; continuously grazed	0.99-1.50 (Sanderman et al. 2015; Teague et al. 2011)	Higher	Inconclusive	0.98-1.00[c] (Hawkins 2017) (Derner and Hart 2007)	Higher (Hawkins 2017)
Organic grass receiving organic fertilizer	Grassland: receiving synthetic fertilizer	1.20 (Kidd et al. 2017)		Higher (Mueller et al. 2014)	0.70-1.50 (Mueller et al. 2014) (Kidd et al. 2017)	Inconclusive
	Grassland: receiving no fertilizer	1.30 (Gravuer et al. 2019)		0.94 (Gravuer et al. 2019)	1.98 (Gravuer et al. 2019)	Inconclusive

Table 7(continued). Indicative main "on-farm" effects of 16 agroecological practices (expressed as effect of intervention divided by baseline with illustrative references

Agroecological Practice	Counterfactual or baseline	Soil carbon	Biomass carbon	On-farm biodiversity	Mean crop, grass or livestock yield	Input costs
Tree crops	Annual crop production	1.18 (Guo and Gifford 2002)	Higher	Higher or similar (Simon et al. 2010)	0.75-1.60 (Bidogeza et al. 2015)	Inconclusive
Tree intercropping	Annual crop production	1.16 (Kim et al. 2016)	Higher	1.37 (Torralba et al. 2016)	0.42-1.00[d] (Garcia de Jalón et al. 2018a)	Lower to higher (Garcia de Jalón et al. 2018b)
Multistrata agroforestry	Monoculture permanent crops	1.57 (Zake et al. 2015)	Higher	Higher (De Beenhouwer et al.2013)	Variable (Niether et al. 2019)	Inconclusive
Silvopasture	Grassland	1.00-1.18 (Upson et al., 2016; Seddaiu et al. 2018)	Higher	1.21 (Torralba et al. 2016)	0.77-1.18[d] (Seddaiu et al. 2018) (Torralba et al. 2016)	Similar to higher (Garcia de Jalón et al. 2018b)
Rewilding and abandonment of agriculture	Crop and grazing systems	Higher (Conant et al. 2001)	Higher	Variable (Rey Benayas et al. 2007) (Lasanta et al. 2015)	0.11-0.80 (Cerqueira et al. 2015)	Inconclusive

Note that the choice of references are illustrative and their inclusion is based on the subjective assessment of the authors after reviewing a range of papers for each practice.

[a]: Will be higher with inclusion of hedgerows

[b]: For non-pollinated crops

[c]: Whilst grass production may be similar; multi-paddock systems may allow higher stocking rates

[d:]: Crop and grass yield responses in agroforestry are very sensitive to number of trees per unit area;

Positive effect: Positive/similar: Similar or very variable: Similar or negative: Negative:

3.3.1 Crop rotations

The use of crop rotations is a well-established agroecological practice where different crops are grown in sequence on the same arable land. We reviewed seven papers (See Table A.1 in the Appendix). The main impacts are described in Table 8 and below with the quality of evidence indicated in brackets. The use of "break crops" can disrupt the build-up of weeds and soil-borne diseases and there can be nutritional benefits if the break crop is a legume (Angus et al. 2015). The type of break crop is important, for example there is little rotational benefit of growing wheat after wheat compared to wheat to barley (Angus et al. 2015). **An important assumption is that the yield of the break crop is of similar economic importance to the main annual crop**.

Soil carbon: in a global study, Liu et al. (2022) indicated that crop rotations significantly increased soil organic carbon (SOC) in the uppermost 20 cm. They related this to a greater diversity in the form of organic matter added to the soil and greater quantities of biomass production. The use of crop rotations also reduced weed density.

Biodiversity: crop rotation increased soil microbial diversity (Venter et al. 2016), and biodiversity in general (Beillouin et al. 2021).

Yield: a global meta-analysis indicated that the effect of planting different crops in succession on crop yields is positive, and this was attributed to reduced pest, weed and disease pressures (Angus et al. 2015). However there is no yield benefit of growing wheat after another non-wheat cereal (Angus et al. 2015).

Greenhouse gases: the inclusion of a legume crop into a cereal rotation can reduce GHG emissions (MacWilliam et al. 2018).

Evidence gaps: the benefits of crop rotations are predicated on the profitability and usefulness of the break crops. Hence research to increase the usability and gross margins of break crops, such as dried peas, can be particularly fruitful.

Table 8. Impacts of crop rotations relative to continuous arable crops

Statement	Confidence	Effect
Soil carbon: Crop rotations increase soil carbon compared to continuous annual monocrops	Well established	Benefit
Biodiversity: Crop rotation increases soil microbial diversity Crop rotation increases biodiversity	Well established Well established	Benefit
Yields: Inclusion of non-cereal break crop increases yield of subsequent wheat	Well established	Benefit
Inclusion of non-wheat species have no effect on yield of subsequent wheat	Well established	Similar
Greenhouse gases: crop rotation with cereal and legume reduces GHG emissions per ha and per tonne yield compared to monoculture cereal	Established but incomplete	Benefit
Other: Crop rotation reduces weed density	Well established Well established	Benefit Benefit
References reviewed for rotations: Angus et al. (2015), Beillouin et al., (2021). Bowles et al. (2020), Liu et al. (2022), MacWilliam et al. (2018), Venter et al. (2016), Weisberger et al. (2019)		

3.3.2 Conservation agriculture

We reviewed 22 papers that quantified the impact of conservation agriculture or more specifically the effect of no tillage relative to conventional tillage (See Table A.2 in the Appendix). The main impacts are described in Table 9 and below with the quality of evidence indicated in brackets. Because large areas of conservation agriculture depend on the use of glyphosate (Schmitz and Garvert 2012), the possible risk of ban on the use of glyphosate is an area for research.

Soil carbon: the lack of tillage associated with conservation agriculture leads to increases in soil organic carbon in the surface layers (Well established). For example, Haddaway et al. (2017) in a meta-analysis of boreal-temperate regions report a 9% increase in soil organic carbon concentration and stock at a depth of 0-30 cm. However there are reports suggesting the increase in the surface layers could be more than offset by declines in soil organic carbon between depth of 10 to 60 cm, due to slower incorporation of crop residue into this soil layers under no-tillage (Cai et al. 2022). Soil organic carbon content below 60 cm was assumed to be similar for both systems. Cai et al. (2022) therefore argues that there is no net benefit of no tillage compared to intensive tillage during the first 10 years of implementation.

Biodiversity: Doran (1980) reports that the level of soil biodiversity in the top 7 cm of soil increased with no-tillage, but that it decreased below 7 cm (Established but incomplete).

Table 9. Impacts of conservation agriculture, and specifically no-tillage (NT) relative to conventional tillage (CT)

Statement	Confidence	Effect
Soil carbon: NT, relative to CT, increases soil carbon in surface layers	Well established	Benefit
NT, relative to CT, reduces soil carbon at depths between 10 and 60 cm	Established but incomplete	Disadvantage
NT and CT have similar soil carbon contents below 60 cm	Established by incomplete	Similar
Biodiversity: NT, relative to CT, increased diversity in surface layers but decreased it at depth	Established but incomplete	Similar
Yields: NT and CT result in similar mean yields of oilseed and cotton	Well established	Similar
NT and CT results in similar mean yields of maize and wheat under dry unirrigated conditions	Well established	Similar
NT, compared to CT, reduces mean yields of root crops	Well established	Disadvantage
NT, compared to CT, reduces mean yields of maize and wheat when there is no or minimal drought stress	Well established	Disadvantage
Costs: NT, relative to CT, reduces fuel costs	Well established	Benefit
NT, relative to CT, increases farm profitability	Inconclusive	
Other: NT and CT have similar greenhouse gas emissions per unit food	Unresolved	
References reviewed for no-tillage: Alluvione et al. (2009); Bayer et al. (2015); Blanco-Canqui and Lal (2008); Cai et al. (2022); Doran (1980); Drawdown (2017); Fernandez (2016); Haddaway et al. (2017); Huggins and Reganold (2008); Hutchinson et al. (2007); Mathew et al. (2012); Metay et al. (2009); Passianoto et al. (2003); Pittelkow et al. (2015); Potter et al. (1997); Robertson et al. (2000); Roldán et al. (2004); Smith et al. (1998); Tuomisto et al. (2013); VandenBygaart et al. (2003); West and Post (2002)		

Yield: Pittelkow et al. (2015) in a global meta-analysis reports that conservation agriculture results in mean yields that were 86% to 101% of those obtained with tillage. They reported similar yields for oilseeds, legumes and cotton, and under dry conditions for maize and wheat (Well established). One reason for this is improved soil moisture retention. However in other environments there was typically a yield loss (Well established). Reasons for this include poorer seed-soil contact at establishment and weed control (Giannitsopoulos et al. 2019). Hence, the mean 8% yield benefit of conservation agriculture relative to conventional agriculture quoted by Drawdown (2017) seems high.

Other: there was no consistent reported effect on greenhouse gas emissions (Unresolved), with a tendency for CO_2 emissions to reduce and N_2O emissions to increase. Conservation agriculture typically results in lower machinery and fuel costs associated with no tillage relative to ploughing (Well established). We did not find clear evidence of the effect of conservation agriculture on farm profitability (Inconclusive), but the combination of similar yields with reduced costs means that it is financially profitable in some places. In fact in many regions, conservation agriculture is now viewed as "conventional" agriculture (Pretty 1995, page 208).

Evidence gap: the effect of conservation tillage, relative to conventional tillage, on soil carbon at different soil depths and net GHG emissions, and their permanence over time, remains an area of research.

3.3.3 Cover cropping

We reviewed 20 papers focussed on the impacts of cover cropping (See Appendix A.3). In general, studies distinguish between cover cropping with legumes (which can increase soil nitrogen) and non-leguminous plants. Interest is also growing in the use of multi-species mixtures.

Soil carbon: global meta-analyses have demonstrated that cover cropping (with legumes and non-leguminous plants) can substantially increase soil carbon, compared to bare fallows, typically within a three year time frame (Abdalla et al. 2019, Morugán-Coronado et al. 2020, Jian et al. 2020).

Biodiversity: cover cropping results in similar or greater plant biodiversity compared to a bare fallow (Guzmán et al. 2019) and increases or changes in fungal biomass (Drost et al. 2020, Murrell et al. 2020). Cover cropping can also suppress weed growth (Osipitan et al. 2019). Some experiments show no effect on arthropods and earthworm communities (Fiorini et al. 2022)

Table 10. Impacts of cover cropping (CC) relative to bare fallow soils

Statement	Confidence	Effect
Soil carbon: CC increases soil carbon compared to bare fallow	Well established	Benefit
Biodiversity: CC results in similar or increased biodiversity (plant, macrofauna and mesofauna) compared to bare fallow	Established but incomplete	Benefit
Yields: CC with legumes rather than a bare fallow can result in higher yields, CC with non-legumes can result in similar yields	Established but Incomplete	Benefit Similar
Costs: CC has high establishment costs which reduces gross margin	Established but incomplete	Disadvantage
Greenhouse gas: CC has similar greenhouse gas emissions as non-CC	Established but incomplete	Similar
Other: CC with non-legume decreases nitrate leaching CC reduces soil erosion compared to bare fallow CC reduces groundwater recharge relative to bare fallow	Well established Well established Established but incomplete	Benefit Benefit Disadvantage
References reviewed for cover cropping: AHDB (2020); Abdalla et al. (2019); de Baets et al. (2011); Drost et al. (2020); Fiorini et al. (2022); Guzmán et al. (2019); Haruna et al. (2020); Jian et al. (2020); Marcillo and Miguez, (2017); Meyer et al. (2019); Miguez and Bollero (2005); Morugán-Coronado et al. (2020); Muhammad et al. (2019); Murrell et al. (2020); Osipitan et al. (2019); Poeplau & Don, (2015); Prechsl et al. (2017); Storr et al. (2019); Thapa et al. (2018); Tonitto et al. (2006)		

Yields: the effect of cover crops on yield is variable and can be confounded by differences in fertiliser management between the treatment and the control (Tonitto et al. 2006). A major global meta-analysis suggested a mean yield decline of 4% using cover crops, but with mixed legume-legume cover crops providing a yield benefit ($p < 0.01$) of 13% (Abdalla et al. 2019). A similar response of similar yields with non-legumes and higher yields with maize with legumes in North America is also reported by Miguez and Bollero (2005) and Marcillo and Miguez (2017). The reduced yields reported by Fiorini et al. (2022) on maize in Italy seem related to compaction issues.

Costs: cover crops have high establishment costs which in a UK study reduced the annual gross margin of the field by £150 per hectare (AHDB 2020).

Greenhouse gases: Prechsl et al. (2017) used an LCA analysis to suggest that there was no significant effect of adding cover crops to the GHG emissions of an arable rotation in Switzerland. Some detailed soil measurements of CO_2 and N_2O emissions are reported by Muhammad et al. (2019).

Other: cover cropping decreases nitrate leaching for non-leguminous plants relative to a bare fallow (Thapa et al. 2018; Tonitto et al. 2006). Cover cropping can also decrease soil erosion (de Baets et al. 2011, Haruna et al. 2020) and ground water recharge (Meyer et al. 2019).

Evidence gaps: there have been some initial attempts to use models to predict how the effect of cover crops on soil carbon will develop over say 60 years (Poeplau and Don 2015). Storr et al. (2019) surveyed UK farmers about their perception of cover crops, with the first and third highest responses received for the positive effect on soil structure and earthworms. These effects were not specifically identified within the papers studied in the rapid evidence review.

3.3.4 Organic crop production

Management: a European meta-analysis by Tuomisto et al. (2012) found that organic, compared to non-organic, farms apply a higher level of organic amendments (Table 11).

Soil carbon: Across many systems, organic agriculture results in a higher level of soil organic carbon (Well established) but applying chemical fertilizer (Han et al. 2016) increases soil organic carbon relative to adding no fertiliser (Well established) (Box 1; Table 11; Table A.4 in the Appendix).

Box 1: Organic amendments and chemical fertilizers both increase soil carbon relative to no addition

Levels of soil organic matter depend on inputs either from plants or animal manure. Greenland et al. (1997) reported that nutrients removed by a crop need to be replaced in some way and that any other approach will be a "dangerous illusion". Smaje (2018) notes that "anecdotal claims that crops will do better without synthetic fertiliser…have to stay on amber until more quantitative data is forthcoming". Our review demonstrates that the overall effect of adding organic amendments (compared to no amendment) is to increase soil organic matter levels. A meta-analysis by Han et al. (2016) indicates that adding chemical fertilizers (compared to no fertilizer) generally increased soil organic matter, due to increased dry matter production. However over a period of time, although adding fertiliser is better than adding no fertiliser, the soil organic matter below arable crops can still decline due to cultivation and the enhanced activity and respiration of soil organisms (Khan et al. 2007). Van Groenigen et al. (2017) also note that a global drive to increase soil organic carbon will need increased levels of soil nitrogen. Syers (1997) argues that in most cases both inorganic and organic inputs are beneficial.

Biodiversity: Studies such as Bengtsson et al. (2005) and Lichtenberg et al. (2017) have demonstrated that organic systems increase the on-farm diversity of birds, soil invertebrates, and arthropods including pollinators (Well established). However in terms of crop yields this also includes the presence of weeds (Well established). We did not find evidence of the effect of adding organic amendments on the biodiversity of non-fertilised cropland (Inconclusive).

Yields: studies such as Cooper et al. (2016) and Clark and Tilman (2017) demonstrate that organic crop production generally results in yields between 48% and 92% of those achieved in well-managed conventional farming systems well-supplied with nutrients (Well established) (Table 11). At a national level, Smith et al. (2018) modelled the effect of an immediate conversion of all agriculture in the UK to organic production. They predicted a change in the product mix and that the total national food output, in terms of metabolisable energy, would be 64% of that under conventional farming. Nitrogen is typically the limiting nutrient in organic systems (Seufert et al. 2012) and Connor (2018) argues that the yield penalty can be larger if there is a need to include nitrogen-fixing legumes (which would otherwise not be required) within a rotation. Such yield penalties contrast with the 8% benefit of converting from conventional arable cropping to regenerative agriculture assumed by Drawdown (2017) derived from three unspecified sources.

The counterfactual is important in describing the yield response. The addition of manure and organic amendments can increase crop yields compared to fields where no other nutrients and amendments are added (Well established; e.g. Pretty, 1995; Tonitto and Ricker-Gilbert 2016), such as in sub-Saharan Africa where in 1996 most soils were losing the equivalent of 22 kg N and 17 kg P per hectare per year (Vlek et al. 1997). However, even in developing countries, the yield loss in organic systems, relative to generally high input conventional systems, can still be large (Seufert et al. 2012). A recent meta-analysis of data from Europe indicated that adding organic amendments increased the yields of some crops such as potatoes and maize under non-nutrient stress conditions, but other crops such as winter-sown cereals did not show a benefit (Hijbeek et al. 2017).

Table 11. Impact of organic crop systems (OS) relative to non-organic systems (non-OS)

Statement	Confidence	Effect
Management: OS tends to receive higher organic inputs than non-OS	Established but incomplete	
Soil carbon: OS tends to have higher soil carbon levels than non-OS	Well established	Benefit
Chemical fertiliser increases soil carbon relative to adding no fertilizer	Well established	Benefit
Biodiversity: OS have higher levels of abundance and species richness of birds, soil organisms, and arthropods than non-OS	Well established	Benefit
OS have higher levels of weeds than non-OS	Well established	Disadvantage
Effect of adding organic amendments to nutrient-stressed crops	Inconclusive	
Yields: OS are lower than those of well-fertilised non-OS	Well established	Disadvantage
Adding organic amendments increases yields of non-fertilised crops	Well established	Benefit
Under non-nutrient stress conditions, adding organic amendments increases potato and maize yields	Established but incomplete	Benefit
Under non-nutrient stress conditions, adding organic amendments resulted in similar yields for winter cereals	Established but incomplete	Similar
Other environmental: OS and non-OS has similar GHG emissions per unit food	Unresolved	
OS and non-OS have similar nitrate leaching per unit area	Unresolved	
Economic: OS uses less energy per unit hectare than non-OS	Well established	Benefit
OS have higher labour requirements and costs than non-OS	Well established	Disadvantage
OS provide lower margins if there is no premium for the product	Well established	Disadvantage
OS provide higher margins than non-OS if there is a premium	Well established	Benefit
References for organic crop systems: Abeliotis et al. (2013); Aguilera et al. (2013); Bengtsson et al. (2005); Clark and Tilman (2017); Cooper et al. (2016); Crowder and Reganold (2015); Diop (1999); Drawdown (2017); Drinkwater et al. (1998); Elshout et al. (2014); Gomiero et al. (2011); Han et al. (2016); Hanson et al. (1997); Hijbeek et al. (2017); Kamenetzky and Maybury (1989); Knudsen (2011); Korsaeth (2012); Kramer et al. (2006); Lichtenberg et al. (2017); LaCanne and Lundgren (2018); Lesur-Dumoulin et al. (2017); Lin et al. (2017); Metcalfe and McCormack (2000); Mondelaers et al. (2009); Ponisio et al. (2015); Rahmann (2011); Robertson et al. (2000); Seufert et al. (2012); Skinner et al. (2014); Tonitto and Ricker-Gilbert (2016); Tuomisto et al. (2012); VandenBygaart et al. (2003); Ziesemer (2007).		

Other environmental: the effect of organic agriculture (compared to non-organic agriculture) on net greenhouse emissions per hectare tends to be more positive when expressed per unit area rather than per unit food, because of the generally lower crop yields. However a meta-analysis by Clark and Tilman (2017) suggests that the overall effect of organic agriculture on net greenhouse emissions per unit food is generally similar to non-organic farming, with some studies showing benefits and some disadvantages (Unresolved). The net effect of organic, relative to non-organic, agriculture on nitrate leaching, eutrophication, and acidification is also largely unresolved.

Economic: meta-analyses such as Clark and Tilman (2017) indicate that organic, relative to non-organic practices, require less energy per unit food and increase the energy-use efficiency of agriculture (Well established). This is primarily by avoiding the use of synthetic fertilisers, as energy use can increase in organic systems. There is also evidence (e.g. Crowder and Reganold 2015) that organic systems require more labour than non-organic systems (Well established). The meta-analysis by Crowder and Reganold (2015) indicates that organic agriculture leads to reduced profitability if there is no organic premium for the final product. However where there is a premium, this is generally sufficient to overcome the shortfall with the effect that most organic systems are more profitable (Well established).

3.3.5 *Integrated pest management*

The Food and Agriculture Organization (2018b) defines Integrated pest management (IPM) as a "careful consideration of **all** available pest control techniques and subsequent integration of appropriate measures that discourage the development of pest populations and keep pesticides and other interventions to levels that are economically justified and reduce or minimize risks to human health and the environment". Integrated pest management (IPM) is supported by the EU directive on sustainable use of pesticides (2009/128/EC). Within IPM, ecological and physical methods are meant to be preferred to chemical methods if they provide satisfactory control (Boller et al. 2004). However Deguine et al. (2001) report, based on research in France, that in practice agrochemical use is still the basis of pest control on most farms practising IPM and hence they argue the case for the specific term of "agroecological crop protection".

In a UK survey (Bailey et al. 2009), the two main practices implemented as IPM were crop rotations and field margin practices such as wildflower strips (which are covered as specific practices elsewhere in this report). Other IPM practices, practiced by more than half the respondents in a UK survey included the timeliness of operations, selection of varieties and variety mixes, hand rogueing, rotation of pesticides, spot spraying, and cultivation to control weeds (Bailey et al. 2009).
Soil carbon: No data was found on the effect of IPM on soil carbon, although cultivation to control weeds could be expected to reduce soil carbon.
Biodiversity: Selection of a greater diversity of cultivars is one aspect of IPM (Bailey et al. 2009). IPM practices that modify the cropped environment (trap crops, pheromone, mixed varieties and introductions) are positively correlated with reduced insecticide applications (Bailey et al., 2009), and reduced agrochemical applications can increase pollinator numbers (Pecenka et al. 2021). Use of physical barriers (e.g. netting and polythene sheets) can have negative effects on pollinators (Egan et al. 2020).

Table 12. Impacts of integrated pest management (IPM) (excluding field margins and rotations) compared to agrochemical-focused pest management

Statement	Confidence	Effect
Soil carbon: no evidence found.	Inconclusive	Unclear
Biodiversity: Increasing cultivar diversity is an aspect of IPM Reduced agrochemical use can increase pollinator numbers	Well established	Benefit
Cultivation and barrier methods can reduce pollinators	Inconclusive	Disadvantage
Yields: increased relative to no IPM (Norton and Mullen 1994)	Established but incomplete	Benefit
Agrochemical use: can be reduced (Norton and Mullen 1994)	Established but incomplete	Benefit
Cost: IPM practices incur costs, but they may be cheaper than pesticide and application costs.	Inconclusive	Unclear
References for integrated pest management (IPM): Bailey et al. (2009); Boller et al. (2004); Deguine et al. (2021); Egan et al. (2020); FAO (2018b); Norton and Mullen (1994); Ortega-Ramos et al. (2022); Pickering and White (2021); Pecenka et al. (2021); Waddington and White (2014)		

Note: The above analysis does not consider rotations and field-margin management (where explicitly mentioned) as these practices are examined elsewhere

Crop yields: Across 61 studies in the USA, Norton and Mullen (1994) reported a mean yield increase from using IPM of 11% , a mean reduction in agrochemical use of 15%, and an increase in net returns of 44%. In the USA, reducing insecticide use in a rotation of maize with a crop benefitting from pollinators i.e. watermelon, had benefits for the number of pollinators and thereby crop yields (Pecenka et al. 2021). In a study covering low and middle-income countries, Waddington and White (2014) found that the use of farmer schools across 460 studies (of which 60% promoted IPM) led to an increase in knowledge by 41%, reduced pesticide use, increased mean crop yields by 13%, and net revenue per unit of land by 19%. In an innovative lab in the UK focused on a single practice, larval numbers of cabbage stem flea beetle were reduced by 45-75% in oilseed rape crops when they were mown or sheep-grazed, but the process also decreased yields (Pickering and White 2021, Ortega-Ramos et al. 2022).

Evidence gaps: Meta-analyses of IPM, such as Norton and Mullen (1994) in the USA, seem rare.

3.3.6 *Integrating livestock into crop systems*

The re-integration of crop and livestock production has been suggested as a method to solve challenges of the global food system (Garrett et al. 2017). This integration can occur at field, farm, and regional levels, but the focus of this analysis is at a farm-level (See Appendix Table A.5). It can be useful to consider the integration of livestock into crop systems (Table 13) separately from the integration of crops into livestock systems (Table 14) as the impacts can be different. Bell and Moore (2012) report that closer integration typically requires more attention to management and reduced integrated typically requires an increase in external inputs.

Table 13. Impacts of integrating pasture and livestock into crop systems

Statement	Confidence	Effect
Soil carbon: Integration of pasture into arable crop rotation results tends to increase soil carbon, but results are often temporary or minimal	Established but incomplete	Similar or benefit
Biodiversity: Pasture integrated into crop land increases abundance of bees	Well established	Benefit
Yields: crop yields in integrated crop livestock systems can be similar to those in crop systems without livestock	Unresolved	
Greenhouse gas emissions: integration of cattle on crop farms increase greenhouse gas emissions per hectare	Well established	Disadvantage
Revenue and costs: Fertiliser costs can be reduced Weed control costs in arable crops can be reduced Mixed systems reduce the inter-annual variation in gross margins Potential to produce marketable product from a cover crop	Established but incomplete	Benefit
Costs to manage livestock increase	Well established	Disadvantage
Other: Zoonotic diseases prevent integration of livestock with leafy vegetables	Well established	Disadvantage
References reviewed for integrating pasture into arable crop systems: Bell and Moore (2012); Carvalho et al. (2010); Cooledge et al. (2022); Hilimire (2011), Liebig et al. (2021); Maughan et al. (2009), Morandin et al. (2007); Peterson et al. (2020); Peyraud et al. (2014); Salton et al. (2014); Sanderson et al. (2013); Sekaran et al. (2021); Tamburini et al. (2022); Tracy and Zhang (2008); Willoughby et al. (2022); Zani et al. (2021)		

Soil carbon: the effect of integration of grazed forage crops into an arable farm is generally to increase soil organic carbon (Salton et al. 2014), but results are often temporary or minimal (Cooledge et al. 2022; Zani et al. 2021).

Biodiversity: integration of pasture and livestock into a crop system increases the agricultural diversity of crops, but also the abundance of arthropods (Tamburini et al. 2022) including bees (Morandin et al. 2007). Animal wastes can also increase the microbial diversity of the soil (Peyraud et al. 2014).

Crop yields: in a meta-analysis, Peterson et al. (2020) reported similar crop yields from integrated crop livestock systems compared to crop systems without livestock; whereas the use of grazed winter cover crop increased mean maize yields compared to continued maize production in the USA (Maughan et al. 2009; Tracy and Zhang 2008). Willoughby et al. (2022) report that an organic system without livestock produced more protein per unit area but less fat per unit area than an organic system with livestock.

Greenhouse gas emissions: integrating cattle into crop systems increases GHG emissions per hectare due to the release of methane by cattle (Liebig et al. 2021).

Costs: the integration of livestock into crop systems increases animal husbandry costs, can potentially provide additional revenue, can decrease fertilizer costs and weed control costs (Hilimire, 2011, Peyraud et al. 2014; Sanderson et al. 2013). In the stocking density is reduced, then loss of nitrogen to the environment can be reduced (Sanderson et al. 2013). Mixed systems can also reduce the inter-annual variation in gross margins (Bell and Moore, 2012; Sekaran et al. 2021)

Other issues: one consideration when integrating livestock into crop systems is the availability of animal husbandry skills (Hilimire, 2011). In addition, different livestock breeds may be more suited for an integrated system, than specialised production (Hilimire, 2011). Zoonotic disease impacts of allowing livestock access to leafy vegetables can also create regulatory and food safety concerns.

3.3.7 Integrating crops into livestock systems

There is relatively little information regarding the benefits or disadvantages of integrating crops into livestock systems. In some cases, the integration of crops into livestock systems should provide the opposite effect of "pasture-fed livestock systems".

Table 14. Impacts of integrating crops on pasture and livestock farms

Statement	Confidence	Effect
Soil carbon: integrated crop livestock systems tend to reduce or have similar soil organic carbon contents as permanent pasture	Established but incomplete	Disadvantage
Biodiversity: increasing heterogeneity could increase biodiversity	Inconclusive	Unclear
Yields: Winter grazing of annual crops can increase livestock feed relative to pasture	Established but incomplete	Benefit
Livestock production increases from integrating a crop with mineral fertiliser on degraded grassland	Established but incomplete	
Mixed systems reduce the inter-annual variation in gross margins	Established but incomplete	Benefit
References reviewed for integrating crops on livestock farms: Bell and Moore (2012); Bell et al. (2015); Bonaudo et al. (2014); de Sant-Anna et al. (2017); Dove et al. (2015); Garrett et al. (2017); Powlson et al. (2011); Salton et al. (2014).		

Soil carbon: integration of annual crops into a permanent pasture system tends to decrease (Salton et al. 2014; Powlson et al. 2011) or statistically similar levels of soil carbon (de Sant-Anna et al. 2017).

Biodiversity: White et al. (2019) using models argued that increasing the heterogeneity of productive land could lead to biodiversity gains, but we did not find field-based evidence.

Yields: Research in Australia suggests that introducing a winter feed crop such as wheat or oilseed rape into a pasture-only system resulted in greater sheep grazing days (Dove et al. 2015) and farm revenue (Bell et al. 2015). Integration of a crop with mineral fertilizer has been beneficial for livestock production on degraded grassland in regions of low natural soil fertility e.g. Brazil (Bonaudo et al. 2014; Garrett et al. 2017). Mixed systems can also reduce the inter-annual variation in gross margins (Bell and Moore, 2012).

Evidence gaps: most of the papers reviewed are outside of Europe and there seems to be a lack of replicated comparisons of integrated and specialised systems in UK and the rest of Europe.

3.3.8 Field-margin agri-environment practices

Across 21 papers, the practices reviewed included include conservation headlands (where agrochemical use is reduced), field margins, hedgerows, set-aside, and wildflower strips (See Appendix Table A.6).

Soil carbon: in a modelling exercise, Falloon et al. (2004) predicted that converting arable land to a grass margin and hedge would increase both soil and biomass carbon. In France, Follain et al. (2007) found soil organic carbon to be 25% greater under hedgerows than at the landscape scale where there had been tillage. In the UK, Holden et al. (2019) reported similar SOC levels under hedgerows as in a grassland field, but higher levels than in an arable field.

Table 15. Impacts of field-margin practices

Statement	Confidence	Effect
Soil carbon: Grass strips increase soil carbon compared to arable Hedgerows increase soil carbon compared to arable fields	Well established Well established	Benefit
Hedgerows have similar soil carbon levels as grassland fields	Established but incomplete	Similar
Biodiversity: wild flower strips and hedgerows generally increase arthropod and pollinator species richness, with effect on cropland greatest in simple landscapes	Well established	Benefit
Set-aside may have similar biodiversity effects as semi-natural grassland	Established but incomplete	Similar
Yields: yields of some insect-pollinated crops can increase from use of wild flower strips and hedgerows	Well established	Benefit
Yields of non-insect pollinated crops are assumed to be reduced in proportion to the area used for hedgerow/flower strip	Established but incomplete	Disadvantage
Yields of arable crops are typically reduced next to hedgerows	Established but incomplete	Disadvantage
Biomass carbon: Grass strips and hedgerows increase biomass carbon compared to arable land use	Well established	Benefit
Soil erosion and water quality: grass margins and hedgerows increase retention of agrochemicals, reducing loss to water courses Grass margins can reduce sediment transport in arable fields.	Well established	Benefit
Runoff: hedgerows increase soil hydraulic conductivity compared to neighbouring fields.	Well established	Benefit
References for field margin practices: Baker et al. (2012); Batáry and Tscharntke (2022); Batáry et al. (2011); Batáry et al. (2015); Chiartas et al. (2022); Dennis and Fry (1992); Drexler et al. (2021); Falloon et al. (2004); Follain et al. (2007); Garibaldi et al. (2014); Holden et al. (2019); Kleijn and Sutherland (2003); Kovács-Hostyánszki et al. (2011); Krimmer et al. (2019); Marini et al. (2016); Marshall (2005); Marshall (2008); Marshall and Moonen. (2002); Patty et al. (1997); Pywell et al. (2015); Vickery et al. (2009); Wooton et al. (2000).		

Biodiversity: an early paper by Kleijn and Sutherland (2003) reported that about half of the investigated agri-environmental schemes in Europe (including organic farming) lacked significant positive effects on biodiversity, although specific schemes focused on, for example, one bird species could be successful (Wooton et al., 2000). That study prompted additional research. Batáry et al. (2011) found that agri-environmental schemes were effective in increasing species richness (primarily focused on arthropods) in grassland areas in both simplified and complex landscapes, and in cropland areas in simple landscapes. These analyses include both off-field and within-field practices. A subsequent analysis by Batáry et al. (2015) found that "off-field" agri-environment schemes substantially increased general diversity relative to the control, with the effect being about twice the level (per unit area) of that achieved with in-field practices. In Germany, Batáry and Tscharntke (2022) found that using wildflower strips on 5-15% of a farm growing wheat could increase bee numbers by 2.5-7.2 fold, and Krimmer et al. (2019) report a 3.6 fold increase in the abundance of pollinators in new flower strip compared to established grassland.

A UK study reported by Marini et al. (2016) found that hedgerows increased the abundance of bees and hoverflies. Vickery et al. (2009) highlights that to maximise the effect of grass margins on biodiversity then it is important to plant a range of plant species, and that planted margins can be more effective for birds, on a per area basis, than organic farming. Redhead et al. (2022b) also reported that agri-environment schemes can increase the abundance of granivorous bird species such as chaffinch, linnet, reed bunting, and yellowhammer). Hedgerows can increase the abundance of arthropod predators (Dennis and Fry 1993). However hedgerows and field margins are not beneficial for all species; for example: skylark, lapwing and stone curlew require "whole-field" options, in part due to predators associated with margins (Vickery et al. 2009; Baker et al. 2012), and Marshall (2008) reported that hedgerows can increase the abundance of mollusc pests. Set-aside was found to result in greater species richness of plants and butterflies than arable fields, but similar levels as grassland fields (Kovács-Hostyánszki et al. 2011).

Yield: For crops benefitting from insect pollination, the addition of pollinator habitats can increase yields. Pywell et al. (2015) reported a 35% increase in the yield of winter beans from planting 8% of the farm to pollinator habitats. In the USA, the use of wild flower strips to increase pollinators increased the yield of blueberries by 20% (Garibaldi et al., 2014). However for crops that do not benefit from insect pollination, then the reduced area of cropping is often assumed to result in pro-rata yield losses (Pywell et al. 2015; Batáry and Tscharntke, 2022). Hedges and field margins can be a source of some weeds, and crop yields can be reduced next to hedges (Pywell et al. 2015). However the net effect on yield of well-managed margins is reported to be generally beneficial (Marshall 2005).

Soil erosion and water quality: the use of grass margins at field edges can reduce the risk of soil sediment and agrochemicals entering water courses (Patty et al., 1997; Vickery et al., 2009). However, it should be noted that due to agrochemical and fertilizer capture, hedges can harbour high levels of nutrients (Marshall and Moonen 2002).

Runoff: soil hydraulic conductivity is generally greater within hedgerows than within fields (Holden et al. 2019).

Evidence gaps: the results of this review have focused more on field margins in arable than field margins in grassland systems.

3.3.9 Pasture-fed livestock systems

Pasture-fed livestock describes the practice where ruminants (usually sheep or cattle in a UK context) feed on only pasture and forage, as opposed to a wider mixture of feed types including cereals. The standards developed for Pasture for Life by the Pasture-fed Livestock Association (PFLA) includes the consumption of grass, legumes, brassicas, herbs within pasture leys, arable silage, and the browsing of shrubby growth (Pasture for Life 2021). They also note that the above can be consumed through grazing or as conserved hay or silage. On welfare grounds, PLFA also allows supplementary feeding of pregnant breeding sheep (Pasture for Life 2021). We reviewed 12 papers covering pasture-fed livestock systems (Table 16) (See Appendix Table A.7).

Soil carbon: at a farm-level, increasing the area of pasture relative to arable crops will tend to result in higher levels of soil carbon, because of the reduced level of cultivation. If pasture-fed systems result in more diverse swards then there may be soil carbon storage benefits (Cong et al. 2014).

Biodiversity: recent research by Norton et al. (2022) in the UK reports that pastures on PFLA registered farms are more plant species rich and taller than improved grassland on non-PFLA registered farms in the Countryside Survey (Carey et al. 2008), perhaps because of differences in grazing practice. In New Zealand, McNally et al. (2015) showed that planting diverse pastures can also increase in higher root biomass in the soil than a simpler ryegrass-clover sward. In a previous section of this report, we indicated that integrating crops into grassland systems could increase agricultural diversity, but it is noted that pasture-fed livestock systems can include brassicas and arable silage.

Yields: feeding a grass-only, rather than a grass and concentrate diet leads to reduced liveweights gains at a similar age, and a delay in animals reaching a specified weight (Herron et al. 2021).

Table 16. Impacts of pasture-fed livestock practices relative to grain-fed livestock

Statement	Confidence	Effect
Soil carbon: soil carbon below pasture is greater than crop systems If pasture-fed swards are more diverse (see below) then there may be carbon benefits	Well established Established but incomplete	Benefit Benefit
Biodiversity: PFLA registered pasture had more plant species than non-registered improved pastures	Established but incomplete	Benefit
Production: Pasture-fed livestock show reduced liveweight gain compared to livestock also fed concentrates for a given age and hence it takes longer for livestock to reach a certain weight	Well established	Disadvantage
Greenhouse gases: pasture-fed livestock increases GHG emissions per unit products compared to grain-fed beef	Well established	Disadvantage
Costs: increased proportion of grass rather than cereals in diets reduce costs.	Well established	Benefit
Other: perceived nutrition and health benefits by some customers	Established but incomplete	Benefit
References reviewed for pasture-fed livestock systems: Bhandari et al. (2015); Capper (2012); Clark and Tilman (2017); Cong et al. (2014); Dillon et al. (2008); Herron et al. (2021); McNally et al. (2015); Norton et al. (2022); Pasture for Life (2021); Smith et al. (2013a); Stampa et al. (2020)		

Greenhouse gases: a global meta-analysis (n = 7) indicates that pasture-fed beef resulted in 19% higher greenhouse gas emissions per weight of product than grain-fed beef (Clark and Tilman 2017; Capper 2012). The diversity of the forage can affect the carbon footprint (Bhandari et al. 2015). However, climate and topographic constraints can mean that grass production is produced in some places where arable production is not possible.

Costs: an increasing proportion of grass in the diet can reduce costs of production (Dillon et al. 2008).

Other: in some countries, pasture-fed rather than grain-fed livestock can increase nutritional security by reducing competition for grain (Smith et al. 2013a). Some consumers perceive nutrition and health benefits from pasture-fed rather than grain-fed meat (Stampa et al. 2020).

3.3.10 Multi-paddock grazing

Soil carbon: multi-paddock systems can result in similar (Sanderman et al. 2018) or increased soil carbon (Teague et al. 2011) compared to continuous grazing (Established but incomplete) (Table 17 and Table A.8). However the effects of grazing system are likely to be confounded by the effects of stocking rate and grazing intensity (Abdallah et al. 2018).
Biodiversity: high, rather than low, stocking rates can reduce plant diversity (Hawkins 2017), but we did not find any evidence of a particular effect of grazing system on plant biodiversity (Inconclusive).
Yield: In a global meta-analysis, Hawkins (2017) reports that multi-paddock and continuously-grazed systems result in similar grass yields. In a detailed study, Nordborg (2016) reports that there is no review study that demonstrates the grass or livestock productivity benefits of holistic grazing relative to conventional or continuous grazing. However Teague et al. (2016) argues that in practice farmers practising multi-paddock or organic systems can achieve better results than observed on experimental stations (e.g. Briske et al. 2008) by adapting actual management to conditions. In some situations, stocking rates may be higher in multi-paddock systems (Badgery et al. 2017).
Other environmental: on some sites, multi-paddock systems have been shown to increase the infiltration of water (Teague et al. 2010). Methods to increase the infiltration of water into the soil (Teague 2018), including the use of contour ripping along keylines can also help control and divert runoff (Duncan 2016).
Economic: multi-paddock systems require increased fencing costs and provision of water sources. However the increased interaction between the livestock manager and the livestock whilst incurring a cost can also improve livestock husbandry.

Table 17. Impacts of multi-paddock grazing (MPG) systems relative to continuous grazing

Statement	Confidence	Effect
Soil carbon: MPG relative to continuous grazing results in similar or increased soil organic matter	Established but incomplete	Benefit
Biodiversity: effect of MPG, relative to continuous grazing	Inconclusive	
Yield: MPG relative to conventional grazing results in similar grass productivity	Established	Similar
Other environmental: MPG relative to continuous grazing can increase infiltration rates	Established but incomplete	Benefit
Economic: MPG increases fencing and management costs relative to continuous grazing	Established but incomplete	Disadvantage
References: Badgery et al. (2017); Chen and Shi (2018); Cox et al. (2017); Derner and Hart (2007); Hawkins (2017); Heitschmidt et al. (1982); Mudongo et al. (2016); Park et al. (2017); Sanderman et al. (2015); Teague et al. (2010); Teague et al. (2011); Wang et al. (2016).		

3.3.11 Organic livestock systems

Soil carbon: a meta-analysis by Gravuer et al. (2019) indicates that adding organic amendments to soil increases soil carbon. Kidd et al. (2017) also showed that the addition of farm yard manure can increase the soil carbon of well-fertilized grassland. Organic systems typically use a higher level of legumes and the addition of legumes generally increases soil carbon (Table 18 and Table A.9).

Table 18. Impacts of organic livestock (OL) relative non-organic livestock (non-OL) systems

Statement	Confidence	Effect
Soil carbon: Adding organic amendments increases soil carbon Adding legumes increases soil carbon	Well established Well established	Benefit Benefit
Biodiversity: Adding organic amendments had no effect on biodiversity	Well established	Similar
Yield: OL with the addition of organic amendments can increase the grass yield of unfertilised rangeland	Well-established	Benefit
OL with the addition of organic amendments can reduce, not affect, or increase the grass yield of fertilised grassland	Unresolved	Variable
Other environment: adding organic amendments reduces runoff Adding organic amendments increases nitrate concentrations	Well established Well established	Benefit Disadvantage
Economic: OL reduces energy use compared to non-OL systems	Established but incomplete	Benefit
OL reduces profitability if there is no price premium OL increases profitability if there is a price premium	Inconclusive Inconclusive	
References: Clark and Tilman (2017); Conant et al. (2001); Dalgaard (2013); Gomiero et al. (2011); Hawkins (2017); Mueller et al. (2014); Gravuer et al. (2019); Topp et al. (2007).		

Biodiversity: in the meta-analysis by Gravuer et al. (2019) adding organic amendments resulted in similar levels of native plant communities.

Yield: the effect of organic livestock systems depends on the counterfactual. In rangeland systems receiving no fertilizer adding organic amendments such as farmyard manure will increase grass yields (Gravuer et al. 2019). However if the existing system involves grassland receiving synthetic fertiliser, moving to an organic system can result in lower yields (Mueller et al. 2014) or higher yields (Kidd et al. 2017) depending in part on the current rate of fertiliser application (Unresolved).

Other environmental: adding organic amendments can reduce runoff but can increase the nitrate concentrations of runoff (Gravuer et al. 2019).

Economic: organic, compared to non-organic, systems generally result in reduce energy use per unit of food (Gomiero et al. 2011). In the absence of specific literature on profitability, we anticipate that organic livestock shows similar profitability characteristics as organic crop production, where profitability depends on a price premium. For example, Duncan (2016) reports that a regenerative agricultural system at Taranaki Farm in Australia depends on direct relationships with consumers and associated premium sale prices.

3.3.12 Tree crops

Our assumption is that new areas of tree crops are grown on existing areas of annual crop production. We reviewed six papers (Table 19; Table A.10).

Soil carbon: soil carbon under tree crops can be greater than that achieved with annual crop production (Guo and Gifford 2002), but the actual level of response will depend on the soil management regime which can range from regular tillage to the use of cover crops (Vicente-Vicente et al. 2016) (Established but incomplete). For example, vineyards can be susceptible to soil erosion (Maetens et al. 2012).

Table 19. Impacts of tree crops relative to arable cropping (AC) or grassland (GL)

Statement	Confidence	Effect
Soil carbon: Tree crops increase soil carbon relative to AC but can vary according to soil management.	Established but incomplete	Benefit
Tree crops have similar levels of SC as grassland	Unresolved	
Biodiversity: Tree crops increase biodiversity relative to AC	Established but incomplete	Benefit
Yields: Tree crops increase calorie production relative to AC	Unresolved	
Tree crops decrease protein production relative to AC	Unresolved	
Other environmental: Tree crops increase above-ground carbon storage relative to AC	Well established	Benefit
Tree crops have similar N_2O emissions compared to AC	Established but incomplete	Similar
Economic: Tree crops increase profitability relative to AC	Established but incomplete	Benefit
References: Bidogeza et al. (2015); Guo and Gifford (2002); Kim et al. (2016); Mutuo et al. (2005); Simon et al. (2010); Vicente-Vicente et al. (2016)		

Biodiversity: Simon et al. (2010) argue that orchards contribute to biodiversity, relative to other arable systems, because of their permanency and multi-strata design. However even organic orchard systems can receive high levels of pesticide application (Katayama et al. 2019). At a global scale, the biodiversity effects of planting tree crops, for example coffee or oil palm, on existing primary forest land is negative (Philpott et al. 2008; Fitzherbert et al. 2008).

Yield: the effect of tree crops on yield is dependent on the specific perennial crop and the baseline arable crop. For example a modelling study in Rwanda (Bidogeza et al. 2015) indicated that bananas increased the calorie production and reduced the protein production relative to maize.

Other environmental: tropical tree crops will increase above-ground carbon storage relative to arable systems (Table 19). Kim et al. (2016) report that a plantation of tropical staple trees did not have a significant effect on nitrous oxide emissions (Established but incomplete).

Economic: a study in Rwanda (Bidogeza et al. 2015) indicated that bananas resulted in greater margins than maize, but that they also required greater labour input and investment.

Evidence gap: we reviewed relatively few examples of the effect of growing tree crops relative to arable or grassland crops in temperate areas.

3.3.13 Tree-intercropping

Tree-intercropping, also known as silvoarable agroforestry and alley cropping, refers to the integration of trees with arable crops.

Soil carbon: there is evidence that tree intercropping systems increases soil carbon levels relative to conventional arable cropping, primarily in the uncultivated areas next to the trees (Established but incomplete) (Table 20 and Table A.11 in the Appendix).

Table 20. Impacts of tree intercropping (TI) relative to arable cropping (AC)

Statement	Confidence	Effect
Soil carbon: TI increases soil carbon relative to arable cropping (AC)	Established but incomplete	Benefit
Biodiversity: TI increases biodiversity relative to AC	Well established	Benefit
Yield: High tree density TI decreases arable yields compared to AC	Well established	Disadvantage
Low tree density TI may result in similar crop yields compared to AC	Established but incomplete	Similar
Other environmental: TI increases above-ground carbon relative to AC	Well established	Benefit
TI reduces soil erosion losses relative to AC	Well established	Benefit
TI and AC results in similar GHG emissions	Unresolved	
TI reduces soil nitrate losses relative to AC	Well established	Benefit
Economic: TI increases labour and management costs relative to AC, assuming continued arable production	Established	Disadvantage
TI can increase or decrease farm profitability relative to AC	Established but incomplete	Similar
TI can result in greater societal values than AC	Established but incomplete	Benefit
References for tree intercropping: Aertsens et al. (2013); Asbjornsen et al. (2013); Garcia de Jalón et al. (2018a); Garcia de Jalón et al. (2018b); Kanzler et al. (2018); Kim et al. (2016); Lin et al. (2017); Thevathasan et al. (2016); Torralba et al. (2016); Tuomisto et al. (2013)		

Biodiversity: a review of European tree intercropping studies has indicated a positive effect on biodiversity relative to arable cropping (Well established).

Yield: there is a wide range of tree-intercropping systems: those with closely-spaced trees will eventually reduce understory crop yields as the tree canopy develops (Well established); however some widely-spaced arrangements where, for example, the arable crop benefits from reduced wind speeds (e.g. Kanzler et al. 2018) may sustain yields (Established but incomplete) (Table 20).

Other environmental: there is strong evidence that tree intercropping increases carbon storage in above- and below-ground woody tissues (Well established). There is mixed evidence as to whether tree-intercropping, relative to arable cropping, reduces net greenhouse gas emissions, as CO_2 emissions generally decrease, but N_2O emissions can increase (Kim et al. 2016). There is modelled and field evidence of reduced soil erosion losses (Well established) relative to arable cropping.

Economic: tree-intercropping typically results in greater labour and management costs than conventional arable cropping, assuming continued arable production (Well established). The relative financial profitability of the system depends partly on the financial return from the tree component ranging from negative (Garcia de Jalón et al. 2018b) to positive effects (Graves et al. 2007). The inclusion of market values for the environmental benefits of such systems typically means that the societal benefit of such systems can exceed that of arable cropping (Established but incomplete).

3.3.14 *Multistrata agroforestry and permaculture*

Soil carbon: a study in Uganda indicates higher soil carbon levels under banana agroforestry than banana monocultures (Zake et al. 2015) (Established but incomplete) (Table 21; Table A.12 in Appendix).

Biodiversity: a meta-analysis by De Beenhouwer et al. (2013) indicates a positive benefit on biodiversity of multistrata agroforestry compared to monoculture plantations.

Yield: the choice of the counterfactual is important when considering the yield of multistrata agroforestry. Whitefield (2011) writes "there's little doubt that well-designed permaculture systems can yield at least as much as conventional high-input systems", but he does not provide quantified evidence. In some situations, multistrata agroforestry will result in a lower crop yield of a specific crop than a monoculture, but total crop production can be higher (Niether et al. 2019).

Table 21. Impacts of multistrata agroforestry (MA) relative to a perennial monoculture (PM)

Statement	Confidence	Effect
Soil carbon: MA relative to PM increases soil carbon	Established but incomplete	Benefit
Biodiversity: MA increases biodiversity relative to PM	Well established	Benefit
Yield: MA, relative to monocultures, can reduce yields of the specified crop, but increase total yield	Unresolved	Variable
Other environmental: MA, relative to PM, increases above ground carbon	Established but incomplete	Benefit
Economic: MA, relative to PM anticipated to increase labour requirements MA, relative to PM, increases farm profitability	Inconclusive Inconclusive	
References: Dal Sasso et al. (2012); De Beenhouwer et al. (2013); Guo and Gifford (2002); Kim et al. (2016); Niether et al. (2019); Ortiz-Rodriguez et al. (2016); Santos et al. (2019); Zake et al. (2015)		

Other environmental: multistrata systems increase above-ground carbon storage relative to monoculture systems (e.g. Niether et al. 2019).

Economic: it is anticipated that multistrata systems will increase labour demands relative to monoculture systems, but this and the effect on profitability were unresolved by our literature review.

3.3.15 Silvopasture

Soil carbon: The overall effect of integrating trees on grassland in a silvopastoral system on below-ground carbon ranges from similar (Upson et al. 2016) to positive effects (Seddaiu et al. 2018) (Established but incomplete) (See Table A.13 in the Appendix and Table 22).

Biodiversity: a European meta-analysis (Torralba et al. 2016) indicates a positive effect of integrating trees on grassland on biodiversity (Established)

Yield: the effect of trees on pasture production depends to a large extent on the number of trees per hectare. High tree densities can supress grass yields, but low densities can enhance production, and can often provide additional fodder. The impact can also be affected by whether the grass is fertilised or not; with the effect of the trees likely to be more positive where the grass is not fertilised (Moreno Marcos et al. 2007).

Other environmental: integrating trees on grassland increases above-ground carbon storage and reduces soil erosion (Torralba et al. 2016) (Well established).

Animal welfare: stakeholders perceive that silvopasture systems improve animal welfare (Garcia de Jalón et al. 2018a).

Economic: the inclusion of trees tends to increase management and labour costs (Well established). The net effect of such systems on farm profitability is unresolved.

Evidence gap: no studies of the effects of silvopasture on greenhouse gas emissions were reviewed for this report.

Table 22. Statements related to silvopasture (SP) relative to grassland

Statement	Confidence	Effect
Soil carbon: SP relative to grassland results in similar or increased below-ground carbon	Established but incomplete	Benefit
Biodiversity: SP relative to grassland increases biodiversity	Well established	Benefit
Yield: the effect of SP on grassland yields depends on the tree density	Established but incomplete	Variable
Welfare: SP relative to grassland increases livestock welfare	Established but incomplete	Benefit
Other environmental: SP relative to grassland increases above-ground carbon SP relative to grassland reduces soil erosion	Well established Well established	Benefit Benefit
Economic: SP relative to grassland increases farm labour SP relative to grassland increases farm profitability	Well established Unresolved	Disadvantage
References: Aertsens et al. (2013); Costa et al. (2018); Garcia de Jalón et al. (2018a); Seddaiu et al. (2018); Moreno Marcos et al. (2007); Torralba et al. (2016), Upson et al. (2016)		

3.3.16 Rewilding and land abandonment from agriculture

In this report, rewilding is defined in terms of naturalistic grazing with relatively passive management. By contrast, agricultural land abandonment can refer to where land has not been converted to forestry or artificial areas and there is total cessation of agricultural activities (Castillo et al. 2021). We reviewed 12 papers (Table 23; Table A.14).

Soil carbon: it is generally considered that rewilding and land abandonment results in increased soil carbon due to the lack of tillage and greater coverage of perennial plants (Lasanta et al. 2015).

Biodiversity: the effect of rewilding and land abandonment on biodiversity depends on the counterfactual (Queiroz et al. 2014). Abandonment of extensive grazing areas and the establishment of closed forest can reduce long-term biodiversity (Rey Benayas et al. 2007; Lasanta et al. 2015), as well as creating problems with invasive species (Corlett 2016). By contrast including large herbivores in rewilding schemes on agricultural land can prevent canopy closure and enhance biodiversity (Ceausu et al. 2015).

Table 23. Impacts of rewilding and land abandonment relative to conventional crop or grazing system

Statement	Confidence	Effect
Soil carbon: increased by perennial relative to non-perennial vegetation	Well established	Benefit
Biodiversity: Abandonment of extensive grazing can reduce biodiversity Rewilding of intensive arable can increase biodiversity Rewilding can increase presence of invasive species	 Established but incomplete Established but incomplete Established but incomplete	 Disadvantage Benefit Disadvantage
Yields: rewilding reduces food production relative to conventional crop or grazing	Well established	Disadvantage
Other environmental: increased perennial woody vegetation increases above ground carbon	Well established	Benefit
Animal welfare impact of rewilding is debated	Inconclusive	
Economic cost: of restoration has not been reviewed	Inconclusive	
References: Ceaușu et al. (2015); Cerqueira et al. 2015; Conant et al. (2001); Corlett (2016); Guo and Gifford (2002); Lasanta et al. (2015); McLauchlan (2006); Rey Benayas et al. (2007); Silver et al. (2000); Smiraglia et al. (2016); Spencer (2017); VandenBygaart et al. (2003)		

Yield: food production is reduced through rewilding and agricultural abandonment (Smiraglia et al. 2016; Cerqueira et al. 2015), although if the land is already marginal the absolute effect on food production may be small. The Knepp rewilding project across 1100 ha in lowland UK annually results in about 75 tonnes of high value beef, pork and venison (Spencer 2017). Whilst the high value may make the system profitable, the quantity of meat is only about a tenth of that achieved, for example, by typical lowland sheep production (Redman 2018). The meat may be marketed as "pasture-fed" because of the lack of a rewilding standard.

Animal welfare: an important topic related to rewilding is animal welfare. There is a need to establish the extent to which it is necessary to protect animals from "hunger, thirst, discomfort, pain, injury, and disease" (Lorimer et al. 2015).

Other environmental: rewilding and land abandonment will generally increase the level of woody perennials and hence above-ground carbon storage.

Economic: land abandonment from agriculture can be inexpensive. Rewilding schemes are generally less labour intensive than agricultural production but may require up-front investment in terms of fencing (Inconclusive). Rewilding may encourage other non-agricultural sources of income.

3.4 Conclusions

Most of the 16 agroecological practices demonstrated **positive impacts in terms of on-farm biodiversity and/or increased soil and biomass carbon** (Table 7; Table 24) relative to the stated baseline. The biodiversity benefits were derived from an increased diversity of crops, the introduction of plants that attract pollinators, provision of different habitats, reduced grazing pressure, and/or reduced use of pesticides and herbicides. The benefits in terms of soil carbon are due to increased crop cover, the introduction of grass into arable systems, reduced cultivation, and/or the addition of soil amendments in organic systems. The practices which did not show a well-established increase in biodiversity or soil carbon were conservation relative to conventional tillage, and the integration of crops into grassland systems. Whilst conservation tillage shows increased soil carbon in the surface layers, there is some evidence that this is offset by reductions at a depth of 10 to 60 cm (Cai et al. 2022). In general, an increase in soil and biomass carbon are associated with a decrease in GHG emissions, except for pasture-fed livestock where reduced growth rates are associated with higher methane emissions per kg of meat.

Table 24. Summary of the predicted impact of 10 agroecological practices on the growing of continuous wheat and 8 agroecological practices on mixed feed continuously-grazed livestock production, based on the results in this report. Positive responses are shaded green and negative responses are shaded red.

Baseline	Implemented practice	Crop yield[ab]	Produce value/kg	Costs	Bio-diversity	Soil carbon	Biomass carbon	GHG emissions
Continuous wheat	Rotations	↑	=	?	↑	↑	=	↓
	Conservation tillage	=	=	↓	=	?	=	?
	Cover crops	=	=	↑	↑	↑	↑	=
	Organic cropping	↓	↑	=	↑	↑	?	=
	IPM	↑	=	?	↑	?	=	?
	Integrate livestock	?	↑	↑	↑	?	=	↑
	Field margins	↓	=	↑	↑	↑	↑	?
	Tree crops	↓	=	?	↑	↑	↑	↓
	Alley cropping	?	=	↑	↑	↑	↑	↓
	Rewilding/abandon	↓	?	?	↑	↑	↑	↓
Mixed-feed continuously-grazed livestock	Integrate crops	↑	=	?	?	↓	?	?
	Field margins	?	=	=	↑	=	↑	?
	Pasture-fed	↓	?	↓	↑	↑	=	↑
	Multi-paddock	=	=	↑	?	↑	?	?
	Organic livestock	?	↑	=	=	↑	=	?
	Tree crops	↓	=	?	↑	?	↑	↓
	Silvopasture	?	=	↑	↑	?	↑	?
	Rewilding/abandon	↓	?	?	↑	?	↑	↓

[a]: Yield refers to the crop yield in the wheat comparison and ignores livestock and tree products.
[b]: Yields refers to livestock production in the livestock comparison and ignores crop and tree products
Note: up arrow demonstrates an increase; down arrow demonstrates a decrease, = signifies a similar response, and ? indicates the response is unresolved or inconclusive. GHG = greenhouse gas.

Within Table 24, the yields refers to either crop or livestock production depending on the baseline, so growing tree crops instead of arable or grassland systems will lead to a reduction in arable yields or livestock production. In practice, this could be offset by, for example, fruit production. Organic cropping and field margins in arable systems and pasture-fed livestock compared to mixed-feed livestock are predicted to result in production losses. However organic cropping and livestock, with appropriate certification, may result in higher product values per kilogramme. For six of the agroecological combinations there was well established evidence of increased costs. The two practices reported to reduce costs are conservation tillage and pasture-fed livestock, and it is interesting that these two practices seem to be widely practised.

Within Table 24, 47 out of the 126 combinations (37%) were rated positive, and 16 (13%) were ranked as negative. With the exception of crop rotations and field margins in grassland systems, there are very few win-win-win practices in terms of soil carbon, biodiversity, and yields. By contrast when Fabulous Farmers (2021) presented the effect of eight agroecological measures on 10 sustainability metrics, only 5 out of 90 combinations (6%) were ranked as negative (Table 25). A recent paper by Tamburini et al. (2020) also highlights the numerous win-wins of the effect of agroecological practices on ecosystem services without affecting yields. The greater incidence of trade-offs in our study compared to Fabulous Farmers primarily results from the inclusion of some negative yield effects and the inclusion of cost as a metric.

Table 25. Reported impact of nine agroecological measures on 10 sustainability metrics as derived from a recent report (Fabulous Farmers 2021) suggests only negative effects for five combinations.

	Yield	Fert-iliser use	Pesti-cide Use	Bio-diver-sity	Polli-nation	Soil quality	SOC	GHG	Water quality	Flood-ing
Mixed crops/rotations	↑	↓	↓	↑	↑	↑	↑↓	↑↓	↑	↑
Reduced tillage	↑↓	↑	↑	↑		↑	↑↓	↑↓	↓	↑
Sward diversity	↑↓	↓	↓	↑	↑	↑	↑	↑↓	↑	↑
Cover crops	↑↓	↓	↓	↑		↑	↑	↑↓	↑	↑↓
Modify manure	↑	↓		↑↓		↑	↑	↑↓	↓	↑
Organic matter input	↑	↓		↑↓		↑↓	↑	↑↓	↓	↑
Hedgerow	↑↓		↑↓	↑	↑	↑	↑	↓	↑	↑
Field margins	↑↓		↓	↑	↑	↑	↑	↓	↑	↑
Agroforestry	↑↓	↓	↑↓	↑	↑	↑	↑↓	↑↓	↑	↑

Note: green represents positive effects and red represents negative effects. Upward and downward arrow = mixed effects. SOC stands for soil organic content and GHG for greenhouse gasses.

The impact of any practice depends on the assumed baseline. The main responses of the 16 practices are outlined below.

Crop rotations: planting different crops in succession results in yield benefits compared to continuous cropping, with the effect greatest if the crops come from different botanical families. Relative to continuous cereal cropping, the practice also results in higher biodiversity and soil carbon. This example of a win-win-win for yield, biodiversity and soil C perhaps explains why crop rotations are practised on most UK farms.

Conservation agriculture: the reduced machinery costs can make the system financially attractive even if there are small yield penalties. There have been recent reports suggesting that the increase in soil carbon found in the surface layers can be offset by a reduction in soil carbon at 10-60 cm, so the soil carbon benefits of conservation tillage are still being debated. In some countries, there is discussion about the future availability of the herbicide glyphosate.

Cover crops: cover crops increase carbon sequestration and biodiversity relative to a bare fallow. If correctly managed, leguminous cover crops can result in increased yields in the subsequent crop, with non-leguminous cover crops resulting in similar yields. A major disadvantage is the cost of the practice.

Organic crop production: on farms where there is currently no fertiliser use, making use of organic amendments (which still incur some costs) can increase crop yields. On farms, where synthetic fertilizers are used, a move to certified organic production will lead to yield decreases of between 8 and 52%, but Crowder and Reganold (2015) report a typical price premium of 25% can be sufficient to make most organic systems profitable.

Integrated pest management: can lead to yield increases, a mean reduction in agrochemical use, and an increase in net margins. Hence there should be advantages of using IPM on most arable farms.

Integration of livestock into crop systems: the greater use of forage rather than arable crops generally increases soil carbon, and subsequent crop yields can be similar. However ruminant livestock will increase greenhouse gas emissions increase per hectare, and to improve profitability, the increased revenue from animal products needs to exceed increased animal husbandry costs.

Integration of crops into grass-based livestock systems: replacing grass with crops will tend to reduce soil carbon, but crops can provide additional feed, particularly at some times of year.

Field margin practices: establishing grass margins, wildflower strips, and hedgerows remove areas from crop production, but will increase soil carbon relative to arable systems, and new hedgerow will increase biomass carbon. Field margins can reduce soil erosion and the risk of agrochemicals entering water courses. Field margin practices can increase slug, arthropods and pollinator numbers. Whilst some bird species benefit from increased presence of hedgerows, some such as skylark, lapwing and stone curlew do not. Hedges and margins can reduce crop yields of non-pollinated crops, but pollinator habitats can increase yields of pollinated crops such as field beans.

Pasture-fed livestock systems: feeding a grass-only, rather than a grass and concentrate diet, leads to reduced feed costs but also reduced liveweight gains at a similar livestock age, and a delay in animals reaching a specified weight. In turn, this means greenhouse gas emissions per unit product are reported to be higher than livestock also receiving grains. Some recent research in the UK suggests that pastures on Pasture Fed-Livestock Association (PFLA) registered farms were generally more plant species rich than improved grassland on non-registered farms.

Multi-paddock system experiments have not demonstrated a grass yield benefit compared to continuous grazing, but the increased management options can allow greater stocking densities and adaptive management. The diversity of grass growth stages should increase habitat diversity, but costs can be greater than continuous grazing.

Organic livestock systems because of the recycling of livestock urine and dung, can sustain similar grass yields to some fertilized grassland systems. Adding organic amendments such as farm yard manure (which will incur a cost) can substantially increase grass yields where there is no mineral fertilizer use.

Tree crops: growing perennial crops on arable land can increase food production and above ground carbon storage.

Tree intercropping: high tree densities will eventually result in lower understorey crop yields as the tree develop, but low tree densities may result in similar yields. The financial attraction of the practice is increased if the tree can also produce financially viable products.

Multistrata agroforestry can offer yield benefits compared to monoculture permanent crops under less-optimal environments, but labour requirements are likely to increase.

Silvopasture grass yields, and thereby livestock production, can be maintained where the tree density is not excessive, and the system can offer animal welfare benefits.

Rewilding and agricultural land abandonment: generally increases soil carbon and can increase biodiversity. Food production will typically be very low, but some areas may have been producing little food before rewilding or abandonment.

It should be noted that the above assessments concern responses at a farm-level. This is similar to a recent study on the effects of agroecological practices in the UK on greenhouse gas emissions (Albanito et al. 2022). The implications of such practices beyond the farm level, depends in part on the assumptions made. For example, if the assumption is that say reduced yields on an individual farm leads to greater food imports then consequential life cycle assessment generally predicts that the negative effects of the reduced yield on global carbon storage and biodiversity can be substantial. By contrast, if we assume that the lack of production results in less consumption or less waste, then the global effect will be closer to that indicated by the farm-scale analysis.

4 Opportunities, barriers, and enablers

4.1 Introduction

This section contains an initial review on published evidence on the major opportunities for, barriers to, and enablers of agroecological innovations, technology and actions to improve productivity and sustainability in the UK. It is based on the review of seven papers by Giller et al. (2021), Sinclair et al. (2019), Mottershead and Maréchal (2017), Jordon et al. (2022), Magistrali et al. (2022), Vermunt et al. (2022), and the Sustainable Food Trust (2022). A fuller review of opportunities, barriers and enabler is presented in the work-package 2 report associated with this project (Hurley et al. 2023).

4.2 Opportunities

In a study in the North of England, farmers indicated that regenerative practices could lead to reduced costs of production (Magistrali et al. 2022). Farmers noted that the primary reason for growing cover crops was "to improve soil structure" and to capture nutrients (Magistrali et al. 2022). It is anticipated that supermarkets and supply chains will be a major driver for regenerative practices that promote carbon storage and reduced GHG emissions as the UK seeks to meet its target for net zero GHG emissions by 2050. Farmers in the North of England indicated that these commercial drivers are likely to appear earlier and be more durable than UK Government schemes to modify land management (Magistrali et al. 2022).

4.3 Barriers

Jordon et al. (2022) interviewed 12 sheep and cattle farmers in Northumberland and Devon to look at the reasons for non-adoption of four agroecological practices: rotational grazing, multi-species herbal leys, integrating trees on farms, and integrating livestock into arable rotations. They reviewed the results in terms of eleven potential barriers for adoption identified by Vanclay and Lawrence (1994), and farm geography (Table 26).

That study identified that the most common barriers to the uptake of regenerative agriculture practices were a lack of knowledge, financial risk, and time and labour requirements.

4.3.1 Lack of knowledge

Magistrali et al. (2022) reviewing the experiences of farmers who have implemented regenerative agriculture practices in the north of England, found that 33 out of 43 respondents identified lack of knowledge as a barrier to adoption (Magistrali et al. 2022). The relative balance between a lack of knowledge and no-barriers tended to be greatest for agroforestry, grazing management, and pasture-based livestock, about equal for cover crops, livestock integration, biostimulants, and organic practices, and the lack of knowledge was least cited for crop diversification, no- or minimum-tillage, and integrated pest management (Magistrali et al. 2022).

4.3.2 Financial barriers

The second most common barrier highlighted by Magistrali et al. (2022) was financial risk. For example, the high cost of seed for cover crops was raised as a barrier. In the Netherlands, Vermunt et al. (2022), who identified five main barriers to agroecological systems in the Dutch dairy sector identified poor financial incentives for farmers and a lack of resources to enable experimentation as two of the barriers. The other barriers were a lack of vision for agroecological approaches, a lack of knowledge and the current food system does not recognise the financial value of the benefits provided by agroecology. Giller et al. (2021) also asks how agroecological practices can be economically and socially integrated into agronomic practice.

Table 26. Barriers to the uptake of rotational grazing, herbal leys, integration of trees, or integrating livestock into arable rotations identified by farmers (Jordon et al. 2022; Vanclay and Lawrence 1994)

Potential barrier	Examples
Environmental	
Farm geography	Climate-related constraints on herbal leys
Information	
Conflicting information	Can herbal ley and rotational grazing benefits be achieved from grass?
Intellectual outlay	Need to seek advice on growing herbal leys
Risk of failure	Unknown establishment and productivity of herbal leys
Inability to trial	Mixed farming has low divisibility due to capital outlay
Financial	
Implementation costs	Fencing and water infrastructure for rotational grazing short-term tenancies
Limited economic benefit	Tree planting and long pay-back time;
Infrastructure requirements	Lack of infrastructure for stock on arable farms
Management	
Incompatible with objectives	Pronounced for farmers identifying as food producers or approaching retirement; reduced stock carrying capacity if land used to grow crops.
Loss of flexibility	Grazing constraints on herbal leys. Inconvenience of trees for farming operations and permanence of land use change
Complexity of intervention	Not mentioned directly

4.4 Enablers

Magistrali et al. (2022) indicated that farmers “were interested in basic research that would baseline the current status of their farms and track changes on a regular basis”. This included carbon and “true cost” accounting. These need to address market failures and reform policies that create perverse incentives is also highlighted by Sinclair et al. (2019).

4.4.1 Reducing uncertainty

Knowledge exchange was highlighted as important to support adoption of rotational grazing and herbal leys (Jordon et al. 2022). Magistrali et al. (2022) also indicated that universities should try out riskier strategies and be “honest and open about mistakes and what didn’t work”. Farmers in Northern England expressed an interest in seeing Newcastle University farms having a more active demonstration role producing regular reports on the financial and environmental outcomes of regenerative practices (Magistrali et al. 2022). An advantage of demonstration farms is that they not only provide evidence, but they are also an effective means of transferring knowledge. CHAPS has proposed a “Field Profiler for Regenerative Agriculture” to provide predictions of the impact of management outcomes (Langford and Taylor 2022).

One way of reducing uncertainty regarding agroecological practices is to fund research on the subject. The Sustainable Food Trust (2022) argues that relevant research and innovation could increase organic yields by an average of 20%. Schmutz et al. (2022) provides a useful review of EU research funding on agroecology and recommendations for both formal and participative research. The Practical Farmers of Iowa (2022) was noted by Masgistrali et al. (2022) as a possible approach to promote regenerative practices.

4.4.2 Financial enablers

Financial incentives were identified as an important enabler by Jordon et al. (2022). One potential method to overcome the financial barrier is the plan to pay farmers in England for the completion of soil management practices, associated with regenerative agriculture, within the Sustainable Farming Incentive scheme (Table 27). However Magistrali et al. (2022) reports that the payments were too low for regenerative practices to be financially viable.

Table 27. Soil standards for i) arable and horticulture, and ii) improved grassland as part of a three-year agreement for the Sustainable Farming Incentive Scheme (UK Government 2022a, 2022b)

Level	Arable and horticulture	Improved grassland
Introductory	Complete a soil assessment and produce a soil management plan Test soil organic matter Add organic matter to **all** land during the 3-year agreement, such as through green manures, catch crops or cover crops, straw, a grass, herbal or legume ley. Have over-winter green cover on at least 70% of the land in the agreement £22/ha/yr	Complete a soil assessment and produce a soil management plan Test soil organic matter No more than 5% bare ground over winter £28/ha/yr
Intermediate	As above, but 20% of the cover must include multi-species green cover £40/ha/yr	As above but herbal leys on 15% of land £58/ha/yr

One way to overcome a financial barrier is to enable a premium for products produced using regenerative practices. However this will require “a standardised method of defining and measuring regenerative practices” (Magistrali et al. 2022). However it was also noted that such practices may become the norm for market access. The Sustainable Food Trust (2022) argue that government subsidies should be conditional on the introduction of whole-farm sustainability assessments.

4.4.3 Systemic enablers and path dependencies

Technological innovations are often promoted by a company who can financially gain from increased sales of a piece of equipment or software. By contrast, agroecological practices are typically not patented and hence there is not a financial incentive for someone to promote (Vanloqueren and Baret 2009; Magistrali et al. 2022). Hence, Mottershead and Maréchal (2017) argue that support for agroecology needs a funding model that “does not rely on the creation of intellectual property or a commercial product”. Mottershead and Maréchal also highlighted the success of the network of Chambers of Agriculture in France to allow rapid knowledge sharing, and the introduction of agri-environment measures (in addition to organic farming) where a farmer can be supported to change an entire farm system through integrated farm management (IFM) rather than just the change in practices.

4.5 Summary

The uptake of agroecological practices depends, in part, on the balance between the opportunities that they offer and the barriers to their implementation. The opportunities created by agroecological practices include improvements in soil health and on-farm biodiversity, and in some cases reduced costs. The increasing requirements being placed on farm businesses by supermarkets and supply chains to reduce GHG emissions is a major driver for regenerative practices, and the durability of those requirements is anticipated to be more durable than UK Government schemes to modify land management. In places the barriers to some agroecological practices will be geographical or incompatibility with management objectives. However where these are not constraints, the barriers are often related to uncertainty or financial considerations. Enablers to overcome those barriers include knowledge exchange (particularly as the promotion of agroecological practices is not driven by a producer wanting to sell a product) and financial enablers (with a focus on market mechanisms that differentiate between desired and undesired societal outcomes, and premium products).

5 Tools to model agroecological practices

5.1 Introduction

This section reviews and appraises tools that can be used to model agroecological systems relative to non-agroecological systems in a UK context, including the use of spatial modelling and mapping and consideration of land-use availability and suitability. This section also seeks to identify gaps in modelling capability, potential gaps in data availability, and the UK's ability to monitor and evaluate the national impacts of agroecological systems.

5.2 Challenges of agroecological modelling

The breadth of factors involved in characterising agroecological relative to non-agroecological practices brings significant challenges for modelling. Changes can be considered at spatial scales ranging from field to the farm, landscape, and whole nation (Wezel and Soldat 2009; Bezner Kerr et al. 2021). Changes can also be considered within agricultural systems or across the whole agro-food system (Wezel et al. 2020). The use of models to examine agroecological practices could cover changes in input parameter values (e.g. reduced agrochemicals), changed processes (e.g. novel crop management), or changed target outputs (e.g. higher levels of soil carbon), and could extend to socio-economic issues such as welfare and equitability. All of this complexity means that it will be difficult to model agroecological futures using any single approach.

The various analytical tools available to model agroecological relative to non-agroecological practices can be broadly categorised either as "models" or "modelling frameworks". In this context, it can be useful to define a "model" as a piece of software capable of estimating the impact of change in a set of input predictors (i.e., model parameters) on a single (or several closely related) output response. By contrast, "modelling frameworks" typically consist of a suite of models applied for a common purpose using common input data. Such frameworks may also include functions to derive model parameter values from user inputs, linkages between models, and/or applications for the visualisation of model outputs. We thus use the term 'tool' generically to encompass our definitions of both models, frameworks, and elements within frameworks.

Some of the key steps in modelling the impact of agroecological relative to non-agroecological practices at a UK scale are described in Figure 6. The rest of Section 5 follows the steps in this workflow, identifying the existing approaches and tools that may be used, adapted or repurposed to support the UK's ability to predict, monitor and evaluate the national impacts of agroecological systems, and the areas where there are gaps in our current understanding, data or capacity.

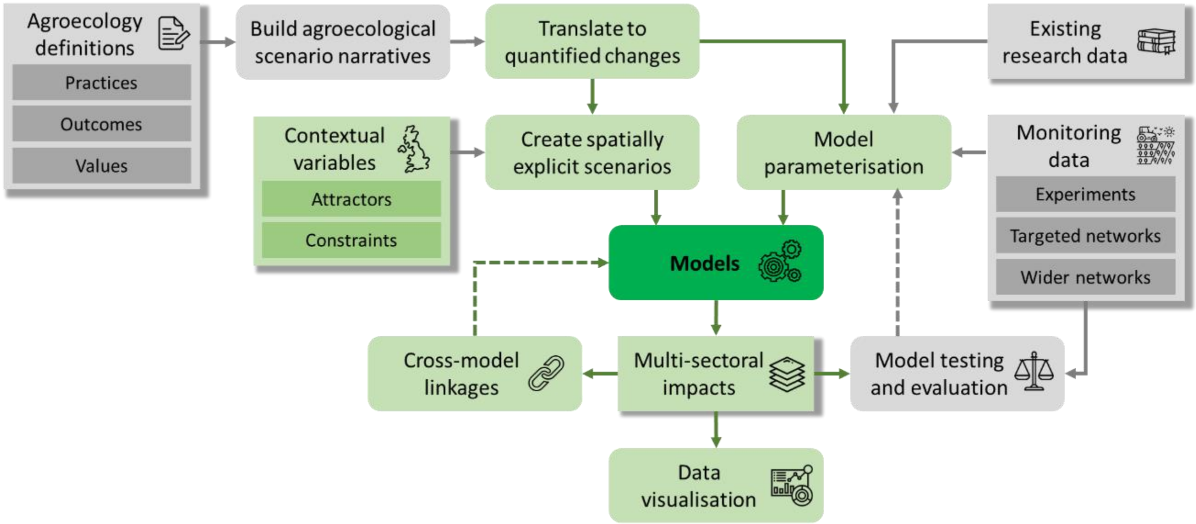

Figure 6. Schematic illustration of steps involved in modelling UK-scale impacts of agroecological relative to non-agroecological systems. Square-edged grey boxes represent data sources and datasets, and round-edged boxes represent processes. Boxes and arrows in green represent those processes and datasets that may be integrated within existing modelling frameworks, whilst grey boxes/arrows represent those that are generally separate from frameworks (as external processes or inputs). Note that, by our definition, 'models' sit within frameworks as the core component translating predictors derived from agroecological scenarios to multi-sectoral impacts, and that there is the potential for iterative feedbacks and linkages (dashed arrows) between models. Icons freely available from Flaticon.com.

5.3 Developing agroecological scenarios

Because the attempt to predict the future exactly is generally futile, the modelling of agroecological relative to non-agroecological practices at scale is usually based on the creation of agroecological scenarios (Audsley et al. 2006). The purpose of scenarios is typically to define a range of possible futures by which to better understand the range of potential outcomes, uncertainties and trade-offs between different elements in the system of interest (Moss et al. 2010, Holway et al. 2012). Scenarios are thus (at their most inclusive level of definition) any form of plausible realisation of future conditions. Types of scenarios can be categorised in various ways, but a common categorisation is to differentiate scenarios according to the type of question which they are attempting to address:

- **Normative** scenarios, also referred to as 'backcasting', are typically used to determine plausible pathways to archive a specific set of targets. For example, normative scenarios could be used to determine pathways to achieve the goal of an agroecological food system or to explore agroecological pathways to achieve a set of independent criteria (e.g. the UN Sustainable Development Goals).
- **Predictive** scenarios typically extrapolate current known trends in order to examine a range of plausible conditions at a defined future time point. Such scenarios usually account for multiple temporal processes simultaneously. For example, in the UK context, predictive scenarios could be used to explore how known drivers such as UK Government policy and climate change affect the extent and impact of agroecological practices (Audsley et al. 2006).
- **Exploratory** scenarios typically ask "what if?" questions based on differing assumptions about the extent, type and location of agroecological change in order to explore a wide range of alternative plausible futures. They also allow the impacts of individual elements of agroecological systems to be explored in isolation or in various combinations.

- In practice, predictive and exploratory scenarios can be applied in combination. For example, climate change projections often provide a core predictive element that interacts with responses to the exploratory aspects (e.g. Audsley et al. 2006; Harrison et al. 2015). In the context of agroecology, normative scenarios are likely to be more challenging to develop, as the definitions of agroecology can simultaneously encompass scenario targets and the practices and pathways used to achieve these (Wezel et al. 2020).

Each of the above scenario approaches could be used to understand the characteristics of a UK food-system including more agroecological practices than present, and their impact on technical, environmental, and socio-economic indicators (Figure 4) relative to a baseline system. In this way, scenarios can be used to provide a range of realisations based on the constraints of a future end-point, current drivers, or management decisions. It is worth bearing in mind that the actual baseline system in the UK includes an existing range of agroecological practices such a cover crops and crop rotations. Although the UK agricultural system is strongly industrialised, farmers still need to consider the constraints placed on them by the environment (Vandermeer 2020) and thus many systems include agroecological practices such as diverse crop rotations, cover crops, and the retention of non-crop habitats in the form of historic landscape features. Additionally, some agroecological practices have received long-term state support, such as conversion to organic or agri-environment schemes (AES), even though few participants using such practices may explicitly identify themselves or their farming systems as ‘agroecological’ (Padel et al. 2020). Ultimately, scenario-based modelling is therefore best placed to provide a range of realisations of how varied agroecological practices and principles, including those elements already extant within the current system (Lacombe et al. 2018; Padel et al. 2020), might combine to create a plausible ‘fully agroecological’ system in the UK context.

When establishing scenarios to determine the effect of agroecological practices in the UK, it can be useful to distinguish between “bottom-up” and “top-down” approaches. Bottom-up approaches assume that key decisions are made at the farm level which then drive patterns that scale up to the landscape (e.g. Audsley et al. 2006; Bohan et al. 2022; Padró and Tello 2022). By contrast, top-down approaches focus on achieving objectives at higher levels of organisation and distributing the required change across spatial units (e.g. Poux and Aubert 2018; Redhead et al. 2020; Mosnier et al. 2022). An attraction of the bottom-up approach is that core decisions about agricultural land use are made by farmers at the farm scale (Audsley et al. 2006). The argument for a top-down approach is that the boundaries of a farm unit are irrelevant to many ecological processes, and that individual farmers have only a degree of control compared to the socio-political organisations that shape the agricultural system (Vandermeer, 2020). In practice, in the context of agroecology which can considered at a range of scales, the construction of a scenario using either top-down or bottom-up approaches should subsequently consider the potential for feedbacks operating in the opposite direction, reflecting the fact that agricultural landscapes are composed of interacting socio-ecological elements at multiple, overlapping levels of organisation (Diogo et al. 2022). Indeed, recent integrated modelling frameworks include both top-down and bottom-up elements and are explicit about the links between them (Harrison et al*.* 2019; 2022).

5.4 Tools for constructing agroecological scenarios

Once a broad scenario narrative has been established, the next step is to convert the narrative into quantifiable changes in land use and land cover (LULC) and agricultural practice (Figure 6). In the UK context, it is typically useful to create spatially explicit scenarios and to use models that can generate spatially explicit outputs (Finch et al. 2021). This is because the potential impacts of many agroecological practices can depend on parameters and processes that vary strongly with the local context (e.g. Woodcock et al. 2016), and because localised or regionalised inequalities in effects can be important from political and pragmatic points of view (Reed et al. 2009).

Tools that have been used to translate a narrative into quantified changes in land use and land cover in the UK include SFARMMOD (Audsley et al. 2006), CLUE (Britz et al. 2011), and the FABLE Calculator (Mosnier et al. 2020). SFARMMOD operates by attempting to maximise per-farm profitability under a given scenario narrative. The FABLE (Food, Agriculture, Biodiversity, Land-Use, and Energy) Calculator (Mosnier et al., 2020; Mosnier et al., 2022) is an open-source Excel-based tool used to study the potential evolution of food and land use systems. The inputs include assumptions such as the demand for agricultural products, current and future diets, and population levels, and outputs quantified LULC change (Smith et al. 2022a). Versions of the FABLE Calculator have been parameterised for the UK (Smith et al. 2022b), and have been successfully downscaled for individual countries (e.g. Wales, Smith et al. 2022a), but do not directly incorporate agroecological practices, being more focussed on top down drivers of LULC change. This situation is common across tools, with some agroecological practices being relatively easy to simulate via existing tools, and others requiring adaptation of the same. For example, tools such as SFARMMOD operate by attempting to maximise per-farm profitability under a given scenario narrative, but the social elements of agroecology may dictate that equitability or efficiency become more important metrics to optimise (Bezner Kerr et al. 2021).

Creating spatially explicit realisations of a given scenario requires knowledge on both how much change is likely to occur (as derived from tools such as FABLE Calculator) and where this change is likely to take place. Within spatial models, the availability, potential, or suitability of a given area for change can be modelled using 'attractors', which increase the likelihood of change in an area, or 'constraints' which reduce or completely preclude the possibility of change (Figure 6). Attractors and constraints may be made up of biophysical (e.g. topography, climate, soils, current LULC) or socioeconomic (e.g. proximity to existing LULC, proximity to supply chain, potential productivity) factors.

Frameworks that use combinations of attractors and constraints to translate scenario narratives into spatially explicit outcomes within a UK context include:

- the Environment and Rural Affairs Monitoring & Modelling Programme Integrated Modelling Platform (ERAMMP IMP), described by Harrison et al. (2022),
- Natural Environment Valuation Online tool (NEVO) described by Day et al. (2019a) and
- ASSIST Scenario Exploration Tool (ASSET) described by Redhead et al. (2020).
- Competition for Resources between Agent Functional Types (CRAFTY-GB, Brown et al. 2022).

An England-focussed equivalent of the ERAMMP IMP called the Environmental Value Assessment Scenario Tool (EVAST) is currently under development. All of these frameworks use a variety of methods to follow pre-set scenario narratives through to spatially explicit realisations and predicted impacts.

There are also examples of standalone tools and datasets outside of these frameworks that can provide a useful source of pre-processed information on agroecological attractors and constraints.

One example is E-Planner (Redhead et al. 2022a). E-Planner provides continuous maps of within-farm and within-field suitability at a 5 m resolution and has been developed to target on-farm environmental management actions (e.g. creation of flower rich pollinator habitats) to areas where the net environmental benefits are potentially greatest. The E-Planner maps are not prescriptive in exactly which management actions are deployed and instead describe the suitability for broad groups of actions with similar attractors and constraints and thus could potentially be a useful baseline on which to build maps of land suitability for agroecology that in turn inform scenarios. A second example is the Agricultural Land Classification. This assesses potential agricultural productivity based on biophysical factors in a way which can be extended to examine change under projected futures (Keay et al. 2012; 2014).

Once attractors and constraints have been identified, collated and mapped, the next stage is to produce a realisation of land use and land cover under a given scenario. This can be achieved by a range of approaches. Simple weighted summation is used by E-Planner (Redhead et al. 2022a)). Multi-criteria optimisation is used in the InVEST rule based scenario generator, which underpins the creation of the scenarios explored in ASSET (Redhead et al. 2020), whilst single criterion optimisation (e.g. of farm productivity is used in the Land Allocation Module of ERAMMP IMP (Harrison et al. 2022) and the agriculture model of NEVO (Day et al. 2019b). CRAFTY-GB takes an agent based approach, where each spatial unit contains multiple agents which compete to determine which land use is best placed to deliver services to meet societal demands set by the scenario narrative, using capitals determined by the attractors and constraints (Brown et al. 2022). Whichever approach is used, in all of these examples, the spatially explicit outputs are largely based on determining changes in broad land use categories such as arable land, improved grassland or forest, driven by societal (e.g. Shared Socioeconomic Pathways) and/or environmental (e.g. climate) change. Although these changes may encompass agroecological practices such as crop diversification, implementation of agroforestry or habitat restoration, many agroecological elements lie outside the scope of current modelling frameworks, so the development of specifically agroecological scenarios are likely to require further development or repurposing of the existing frameworks.

5.5 Selecting models of agroecological impacts

Extant modelling frameworks use a wide range of extensively documented component models (e.g. Day et al. 2019a, Finch et al. 2021, Harrison et al. 2022). Whichever exact models are under consideration, the issues and challenges involved in applying them to agroecological scenarios are likely to be governed by the broad type of model concerned. Individual models may fall into one of three broad categories, each with a variety of advantages and limitations in an agroecological context:

Process based or mechanistic models are reliant on functions simulating biophysical or socioeconomic process. They require mechanistic understanding of the process by which input variables interact to produce the model's output Examples include crop or agroforestry yield models such as DSSAT (Hoogenboom et al. 2019) and Yield-SAFE (van der Werf et al. 2007), which simulate the conversion of solar radiation to plant biomass, mediated by plant physiological responses to variations in temperature, water and nutrient availability. Process based models can be used to predict agroecological impacts outside the range of current conditions and are often sensitive to relatively small changes in model inputs, thus allowing the simulation of subtle changes. However, this sensitivity means that they require accurate parametrisation with data on the impact of agroecological practices on parameter values, which may be challenging to obtain over larger spatial scales. It also means that results can vary widely depending on exactly which process-based model is selected (e.g. Jägermeyr et al. 2021). These models can be computationally intensive to run, limiting their use for rapid exploration of multiple scenarios. Uncertainty can be estimated by varying parameters and exploring the sensitivity of the model to variability in the inputs, to generate a distribution of predicted outcomes under a given scenario.

Statistical models identify statistical relationships between current predictors (e.g. presence of agroecological practices) and the outcome of interest. Examples include species distribution models (SDMs) such as MaxEnt (Phillips et al. 2006), which determine statistical relationships between the binomial probability of species occurrence and spatially explicit predictors. SDMs are frequently used as the biodiversity modelling component of existing modelling frameworks (NEVO: Day et al. 2019a; ASSET: Redhead et al. 2020). Statistical models do not require the existence of mechanistic knowledge by which to simulate the processes linking predictors and outputs, but they do typically rely on the assumption that historic associations are a valid predictor of responses to hypothesised future change. They also require sufficient data on the current spatial patterns of predictors of interest - many agroecological practices are currently taken up at very low levels in the UK (Padel *et al.*, 2020), and there is little consistent data on their location and extent, so building robust statistical models is likely to be challenging. Most statistical models produce quantitative estimates of uncertainty (e.g. confidence intervals).

Benefits transfer models are computationally simpler than either of the above approaches. They do not depend on mechanistic understanding or on statistical associations, but instead identify outcomes associated with particular combinations of input variables (e.g. land cover, soil type, agricultural practice) from existing research, and assume that these outcomes are replicated wherever this combination is encountered. Such models are often described as ‘calculators’ or ‘look-up tables’. They are flexible and require comparatively little data to parameterise (only as much as is required to fill in all cells of the look-up table). However, they still require an existing body of research on the impacts of different agroecological actions in different contexts. It is also difficult to quantify uncertainty with this sort of model.

Because the impacts of agroecological systems can be social, environmental, and economic, and vary in the degree to which we have existing data and mechanistic understanding by which to drive predictive models, any agroecological modelling framework is likely to use a variety of models drawn all three of the categories above, as do extant frameworks (e.g. ASSET, ERAMMP-IMP, EVAST, and NEVO) is likely to draw on a range of models. Although making uniform estimates of uncertainty across models is difficult, assessments of uncertainty can still be derived from both quantitative (e.g. model validation) and qualitative (e.g. modeller certainty) data (Dunford et al. 2015).

Two challenges in modelling agroecological systems are the wide range of spatial scales which may be relevant, and the effect of cross-sectoral feedbacks. Practices that affect ecosystem processes such as pollination and pest control may show influences extending for under 100 m (Woodcock et al. 2016), and are contextually dependent upon the farm system and local landscape (e.g. Karp et al. 2018; Haan et al. 2020). At the same time, agroecological systems encompass multiple socioeconomic aspects operating over far larger scales (Wezel and Soldat 2009; Diogo et al. 2022), which may then influence the context within which finer scale processes take place. The importance of considering cross-sectoral feedbacks was demonstrated by Harrison et al. (2016) who showed that using the CLIMSAVE Integrated Assessment Platform to model cross-sector dependencies and feedbacks predicted substantially different effects on food production, irrigation, proportion of arable land, and carbon storage at a European scale than using single sector models. Given the cross-sectoral and multi-scale nature of agroecology it is highly likely that potentially misleading results will be generated from single-sector models. One approach to developing integrated models with cross sectoral linkages that can run quickly and efficiently is to develop meta-models, which are "computationally efficient or reduced form models that emulate the performance of more complex models" (Harrison et al. 2015; 2019). They may also reduce the data required for parameterisation by removing factors which remain constant under all scenarios or to which the model is less sensitive. Meta-models need to be tested to ensure they can reproduce the effects of their more complex parent models in terms of responses to the changes of interest. Hence it is likely that the exploration of agroecological scenarios in the UK across a wide range of potential indicators of sustainability is likely to be most effectively derived using models drawn from existing integrated frameworks either by upscaling approaches made for individual countries (e.g. ERAMMP IMP, EVAST) or downscaling pan-European frameworks (e.g. CLIMSAVE). Their successful adaptation to agroecological modelling rests on the ability to parameterise these models with accurate data to ensure that they can accurately simulate the impact of agroecological practices and changes.

5.6 Parameterising models of agroecological impacts

Establishing the impact of agroecological practices on sustainability indicators at a field or farm level can be difficult. For practices that are relatively novel or have hitherto only been applied at small spatial scales there is often a limited amount of data on the impacts they are likely to have on the biophysical and ecological properties that models require as input parameters. There are some spreadsheet datasets such as that produced by Jouan et al. (2021a) that describe the effect of several agroecological practises at the farm scale typically using co-efficients connecting practices to indicators. However, from the EU wide literature review used by Jouan et al. (2021b) 49% of the coefficients were derived from expert assessment, and only 2% from peer reviewed studies (Jouan et al. 2021b). The review by Burgess et al. (2018) examined the impact of nine agroecological practices on soil carbon and on-farm biodiversity. This paucity of quantitative data on the impacts of agroecological practices has been noted by practitioners, with farmers raising the scarcity of useful information in the UK context (Padel et al. 2020). This problem becomes even more prominent when we wish to consider the parametrisation of inter-model linkages. In the Bezner Kerr et al. (2021) review of agroecological impacts on food security, studies that examined multiple interacting components of agroecological systems were very much in the minority (69% of reviewed studies examined only one or two components), with studies of the impact of the social components (such as social equitability) entirely absent from the dataset.

Even where empirical studies have taken place, the limited sample sizes involved may make it difficult to adequately parametrise process-based models, build robust relationships using statistical approaches or to assess the uncertainties involved in extrapolating their results to wider-scale uptake using benefits transfer approaches. The problems of parameterising can be illustrated using the example of reduced agrochemical inputs (as an example agroecological practice) on the single output of biodiversity. The practice has an existing proxy (organic agriculture) with a relatively long history. Yet constructing a quantitative model linking the impact of reductions in agrochemical application to populations of a given taxon over larger spatial scales is extremely challenging. It is difficult to use process-based approaches as there are few data on the quantitative relationships between pesticide usage, exposure, hazard and populations outside of the laboratory or field-trial scale. Developing statistical relationships is also difficult because correlations between agrochemical usage (which is not generally available) and species population responses can have biases and limitations arising from the restricted availability of data on agrochemical use (Mancini et al. 2019). Using benefits transfer approaches from, for example, studies focused on the removal of agrochemicals from organic systems is problematic, because their results are often confounded with other factors such as changes in tillage, crop rotation and creation of non-crop habitats (Fuller et al. 2005; Hole et al. 2005), and their transferability limited by the small-scale and isolate context of many organic farms (Fuller et al. 2005). Hence an important first step towards successful modelling of agroecological impacts is likely to involve a comprehensive exercise of collating available data and matching specific agroecological practices to candidate models. Such an exercise should also identify data gaps and deficiencies, which could then be addressed.

5.7 Approaches for monitoring and evaluation

The development of sustainable agroecological practices and systems, like any management process, will benefit from effective monitoring and evaluation. This can occur through experiments, targeted networks, or existing national networks (Figure 6; Figure 7). Experiments can be useful for enhancing our mechanistic understanding of processes and provide vital quantitative data for model parameterisation. Targeted networks can provide data to evaluate model performance and sensitivities over large spatial scales, and national networks can be used to test model predictions and monitor uptake is having the expected effects. Ideally a combination of the three approaches can be useful for sense-checking and maximising the usefulness of the available information (Figure 7).

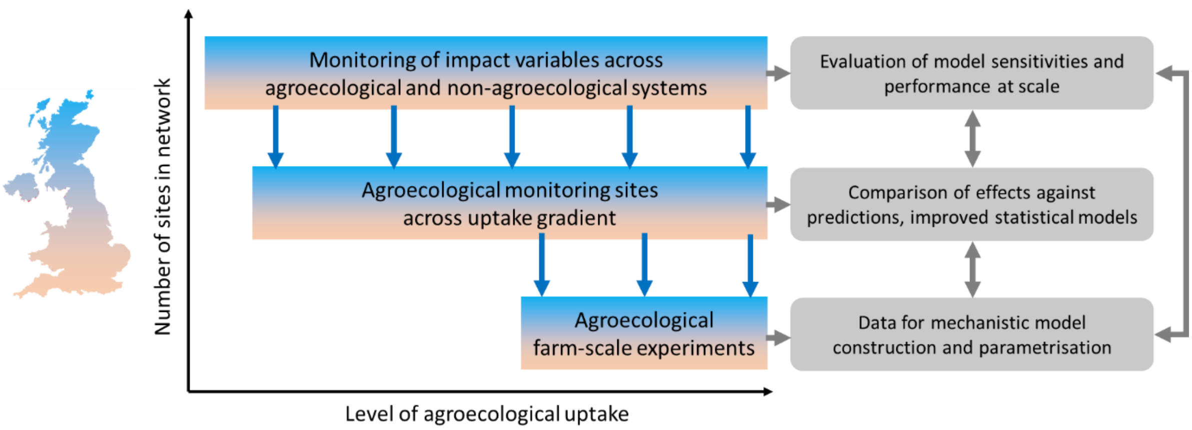

Figure 7. Schematic illustration of three levels of monitoring networks for agroecology (coloured boxes) and their potential use for agroecological modelling (grey boxes). The levels vary in their coverage of the agroecological uptake gradient and the number of sites in the network (as determined by the likely cost and effort in setting up and maintaining the network). The colour gradients indicate the requirement for each network to be representative of a range of UK contexts and conditions. Arrows between boxes indicates the importance of sites common across the three levels of monitoring to explore scalability and transferability.

Individual plot- and field-scale experiments (i.e. the lowest tier in Figure 7) can help improve our mechanistic understanding of processes, which can then be used to improve the effectiveness of process-based models. However to determine the implications of local responses at a national scale, it is useful for the experiments to be sufficiently well-replicated to be representative of a stratified type of farming (see downward vertical lines in Figure 7). This helps to explore transferability of experimental results between contexts or to directly elucidate the relationship between contexts and outcomes so this can be simulated in a model. Such stratified experimental sites were a common feature of early work studying for example the effect of nitrogen on grass yields across different parts of England (Morrison et al. 1980), and there was a UK National Network of Silvopastoral and Silvoarable Agroforestry Experiments (Burgess et al. 2005). One approach for experimental evaluation of agroecological practices is for organisations in the supply chain (e.g. Selvey 2022) or researchers to coordinate a network of farms who are trialling such practices (Catalogna et al. 2018). However, networks created by these 'top-down' approaches are likely to be limited in sample size and replication across spatial contexts. Alternatively, existing farmer-focussed networks (Table 28) could form valuable starting points for evaluating agroecological practices. Although many of these were set up for knowledge transfer and coordination (Prager 2015, Wezel et al. 2018, Rellensmann 2021), product certification (Selvey 2022), some have a remit covering monitoring, experimental work or collation and dissemination of data.

Table 28. Examples of UK-wide networks of farms and farmers with an agroecological or environmental focus

Name of organisation	Website
Farmer clusters	https://www.farmerclusters.com
LEAF Demonstration farms	https://leaf.eco/farming/leaf-network
Nature Friendly Farming Network	https://www.nffn.org.uk/
Agroecology Research Collaboration	https://landworkersalliance.org.uk/agroecology-research-collaboration
Innovative Farmers Project	https://www.innovativefarmers.org/

The effects of agroecological systems could also be evaluated by monitoring sites across a gradient of known levels of agroecological uptake (the middle tier in Figure 7). Although this is similar to the first use case, requiring a substantial network of representative sites, it may not need long-term experimental work. The use of space-for-time analysis techniques may be able to correlate the degree of agroecological uptake to selected sustainability metrics. Repeat monitoring can elucidate this further, to explore whether change in agroecological uptake is correlated with change in impacts. Although this tier of monitoring does not confer the degree of mechanistic understanding of detailed experimental work, it is more plausible to implement at scale, and is potentially able to make use of use a wider range of existing farmer networks (e.g. those in Table 21). However, the challenge is that agroecology encompasses such a wide range of practices that creating a monitoring scheme with factorial combinations of agroecological practices becomes very difficult as each practice is rarely applied in isolation. However, such an approach has been used to evaluate agri-environment schemes, which present a similar challenge to monitoring. A recent agri-environment scheme (AES) monitoring programme constructed generalised gradients of uptake at a 1 km resolution across England, by a sum of individual options weighted by the evidence for their effectiveness and quantity per 1 km square (Staley et al. 2021). Sites for monitoring were then selected to represent high, medium and low levels of the gradient at both local and landscape scales, and to ensure that these gradients were independent of other landscape gradients (e.g. quantity of seminatural habitat). This approach is dependent on the ability to access accurate data on the uptake of AES options. Because accurate data on the configuration and quality of the uptake of the agri-environment scheme is not readily available, this study required extensive field surveys to avoid the limitations faced by previous studies (e.g. Baker et al. 2012; Dadam and Siriwardena 2019) which have had to interpret their results with the caveat that option configuration and quality are largely unknown. In view of the above, ensuring that we can construct gradients of agroecological uptake, and understand how impacts of agroecological practices at an experimental scale to those at a landscape scale will require accurate, standardised data on the uptake and quality of agroecological practices. New technologies and tools at the farm scale, driven in part by the rise of precision agriculture (Bongiovanni and Lowenberg-DeBoer 2004) are likely to help support this, including farmers using digital tools to perform their own mapping of farm practices (e.g. Digital Land Solutions Ltd 2022) or monitoring of wildlife habitats (e.g. UKCEH 2022), alongside increasing awareness within the farm industry of the value of sharing data (Walter et al. 2017).

The third level of evaluation (top tier in Figure 7) could be to use the wide variety of data that is available from existing national monitoring programmes. These include repositories of data from sampling programmes, surveys and citizen science schemes. Even where these data include biases and complexities, statistical techniques can help to unravel these (Mancini et al. 2019), and even where there are limitations, the data can still be useful to evaluate models. For example, UK-wide comparisons of the InVEST water yield and nutrient retention models against measured river flows from the National River Flow Archive and nutrient loads from the Environment Agency Water Quality Archive, showed that the models provides a good prediction of the relative ranking of catchments at a national scale (Redhead et al. 2016, 2018), and thus show potential to provide robust estimates of the magnitude and direction of change under national scale scenarios, even though there was a great deal of uncertainty at finer spatial scales (Gosal et al. 2022). Such evaluation exercises can also be useful in revealing the relative sensitivity of models to changes in inputs and parameters which can inform the construction of effective meta-models and model linkages. However, because national monitoring programmes are not explicitly agroecological in focus, agroecological systems are unlikely to be well represented, and thus there is always a risk that the resultant datasets do not adequately capture a model’s sensitivity to agroecological practices. Thus the linkages across agroecological networks are key (Figure 7) if we are to maximise the amount of information we can derive from monitoring and evaluation programmes.

5.8 Conclusions

Many of the tools required to successfully model the impacts of agroecological relative to non-agroecological systems at the UK scale already exist. The suites of linked, multi-sectoral models used by UK-focussed frameworks (such as ASSET, ERAMMP IMP, EVAST and NEVO) are intended to cover a wide range of impacts encompassing multiple aspect of sustainability and are parameterised for use in a UK context. The gaps in our ability to repurpose these frameworks and their component models for successful agroecological modelling are largely threefold.

First, we must ensure that we can construct plausible agroecological scenarios which we can explore with these modelling approaches. Scenarios need to encompass both ‘top-down’ and ‘bottom-up’ processes involved in determining the impact of agroecological systems. There are tools to translate scenarios narratives to quantitative descriptors (e.g. SFARMOD, FABLE Calculator) and in understating where agroecological transitions are likely to take place (e.g. E-Planner) but determining their relationship to specific agroecological practices or systems would require additional development.

Second, we need to parametrise models with accurate data on agroecological practices. The lack of experimental data (over larger spatial extents and prolonged periods of time) on agroecological practices means that parameterising mechanistic or statistical models is challenging. Conversely simply scaling up farm scale expert-based scoring or benefits transfer approaches can result in predicted impacts with unquantified and potentially large levels of uncertainty. Successful selection and parameterisation of models is likely to involve iterative testing of models, using data collated from existing research and from the establishment of multi-scale agroecological monitoring networks.

Third, and linked to the previous point, it is vital that we are able to test and improve our ability to model agroecological systems if modelling is to be regarded as a useful tool for decision support, whether at the scale of the individual farmer or the setting of national policy. This relies on effective monitoring of the implementation of agroecological practices. Whilst many existing mechanisms are in place within the agricultural sector that may be used to facilitate agroecological monitoring (including precision farming technologies, decision support tools and farmer networks) these need to be brought together with an explicitly agroecological focus to ensure that they are capable of providing data at the required level of openness, accuracy and spatial resolution for model improvement and validation. Whilst the focus of this study has been on bio-economic models, there is also a potential need for improved understanding of the impact of agroecological approaches on social networks.

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Appendix A: Worksheets of evidence

Table A.1. Evidence worksheet for rotations

Inter-vention (A)	Relative to baseline (B)	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/System B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (System A/System B)	GHG emission system A/System B	Weed incidence	Biodiversity	Reference
Rotations													
Rotation	Continuous crop monoculture	Positive impact on soil carbon		Meta-analysis	167	Global			1.06 (0-20 cm)				Liu et al. (2022)
Wheat after barley	Wheat after wheat	No significant yield effect		Meta-analysis	60	Global	1.05						Angus et al. (2015)
Wheat after oats	Wheat after wheat	Positive yield effect		Meta-analysis	150	Global	1.18						Angus et al. (2015)
Wheat after brassica	Wheat after wheat	Positive yield effect		Meta-analysis	180	Global	1.27						Angus et al. (2015)
Wheat after legume	Wheat after wheat	Positive yield effect		Meta-analysis	300	Global	1.37						Angus et al. (2015)
Wheat after fallow	Wheat after wheat	Positive yield effect		Meta-analysis	32	Global	1.34						Angus et al. (2015)
Maize after non-maize crop	Maize after maize	Positive yield effect		Meta-analysis	11	USA	1.28						Bowles et al. (2020)
Rotation of selected crops	Unspecified	Positive effect on yield		Meta-analyses	7	Global	1.16						Beillouin et al. (2021)
Rotation of selected crops	Unspecified	Positive effect on biodiversity		Meta-analyses	2	Global						1.37	Beillouin et al. (2021)
Rotation of more than one annual crop	Single annual crop	Positive effect on microbial species richness		Meta-analysis	26	Global						1.15	Venter et al. (2016)
Rotation of more than one annual crop	Single annual crop	Positive effect on microbial species diversity		Meta-analysis	43	Global						1.03	Venter et al. (2016)
Rotation of more than one annual crop	One crop	Reduced weed density		Meta-analysis	54	Global					0.51		Weisberger et al. (2019)
Cereal after pulse crop (with adjusted N fertiliser)	Cereal after cereal crop	Reduction in GHG emissions per hectare		LCA study	26	North America	1.16			0.35			MacWilliam et al. (2018)
Cereal after pulse crop (with adjusted N fertiliser)	Cereal after cereal crop	Reduction in GHG emissions per tonne		LCA study	26	North America	1.16			0.35			MacWilliam et al. (2018)

Colour code: Positive: Similar Negative: Inconclusive or confounding factors:

Table A.2. Evidence worksheet for conservation tillage

Inter-vention (A)	Relative to baseline (B)	Impact		Type of study	Number of studies	Location	Crop yield ratio: A/B	Additional carbon storage	GHG emission ration (A/B)	Bio-diversity	Labour use	Energy use	Reference
CA	v conv. agriculture			Desk-study		Global	1.08	+0.25-0.71 t C/ha/a	-0.23 t CO2eq/ha/a				Drawdown project (2017)
No-till	v conv. tillage	provides similar yields for oilseeds and cotton in most environments		Meta-analysis	74	Global	1.01						Pittelkow et al. 2015
		provides similar yields for legumes in most environments		Meta-analysis	166	Global	1.00						Pittelkow et al. 2015
		provides similar oat and maize yield in dry unirrigated area		Experiment	1	Brazil	1.00						Bayer et al. (2015)
		increases soil moisture and thereby crop yields in dry environments		Review	1	Canada							Hutchinson et al. (2007).
		reduces yields of root crops in most environments		Meta-analysis	19	Global	0.86						Pittelkow et al. 2015
		reduced yields of maize in most environments		Meta-analysis	224	Global	0.94						Pittelkow et al. 2015
		reduced yields of rice in most environments		Meta-analysis	153	Global	0.96						Pittelkow et al. 2015
		reduced yields of wheat in most environments		Meta-analysis	260	Global	0.97						Pittelkow et al. 2015
							0.97						
No till	v tillage	increases soil carbon in the top 5 cm		Experiment	1	USA		+1% C					Mathew et al. (2012)
No till	v conventional tillage	increases soil carbon in the top 25 cm of soil		Review	14	Europe		+0.71% C/yr					Smith et al. (1998)
		increases soil carbon in the top 30 cm of soil		Review	1	France		+0.1 t C/ha/yr					Metay et al. (2009)
		increases soil carbon in the surface layer		Experiment	1	USA		+0.3 t C/ha/yr					Robertson et al. (2000)
		increases soil carbon in the top 30 cm of soil		Meta-analysis	351	Global		+3.8-4.6 Mg/ha					Haddaway et al. (2017)
		increases soil carbon in the top 15 cm of soil		Meta-analysis	93	Global		0.48 t C/ha/yr					West and Post (2012)
No till	v plough tillage	increases soil carbon in the top 10 cm of soil		Field trials	11	United States		Positive					Blanco-Canqui and Lal (2008)
No Till	v conventional tillage	increases soil carbon in the top 20 cm		Experiment	1	USA		+2.8-5.6 t C/ha					Potter et al. 1997
No Till	v plough tillage	increases soluble soil carbon in top 10 cm		Experiment	1	Mexico		+20 mg/kg					Roldán et al. (2004)
No till	v minimum tillage	had minimal effect on soil carbon in surface layers		Meta-analysis		Western Canada		0					VandenBygaart et al. (2003)
No till	v largely plough	increased soil organic carbon in surface layers		Meta-analysis		Eastern Canada		+2.9 Mg C/ha					VandenBygaart et al. (2003)
No till	v conventional tillage	increased soil organic carbon		Review	1	Canada		+0.05-0.25 Mg C/ha/yr					Hutchinson et al. (2007).
		results in similar levels of soil carbon in 15-35 cm of soil		Meta-analysis	93	Global		0					West and Post (2012)
		results in similar levels of soil carbon in 0-60 cm of soil		Field trials	11	United States		0					Blanco-Canqui and Lal (2008)
		results in similar levels of soil carbon in 0-150 cm of soil		Meta-analysis	351	Global		+0.83-1.65 Mg/ha					Haddaway et al. (2017)
No till	v conventional tillage	reduced N_2O emissions for an oat/maize rotation		Experiment		Brazil			-0.47 kg N/ha				Bayer et al. (2015)
No till	v conventional tillage	increased N_2O emissions in a vetch/maize rotation		Experiment		Brazil			+0.33 kg N/ha				Bayer et al. (2015)
No till	v disc till	resulted in similar N2O and NO emissions		Experiment		Brazil			Similar				Passianoto et al. (2003)
Min-till	v ploughed	tended to increase N2O emissions		Review	19	Mediterranean			+0.9 kh N_2O N/ha/yr				Fernandez (2016) page 97
No tillage	v ploughed	decreased growing season CO_2 emissions		Experiment		USA			-0.33 Mg C/ha				Alluvione et al. (2009)
No till	v disc till	Decreased CO2 emissions		Experiment		Brazil			-2.57 Mg /ha				Passianoto et al. (2003)
No till	v conventional	assumed to decrease GHG emissions per ha in JRC model		European Model		Europe			-0.4 Mg CO_2e/ha/yr				Tuomisto et al. (2013)
No tillage	v ploughed	increased growing season CH_4 emissions		Experiment		USA			+19 g CH_4/ha				Alluvione et al. (2009)
No till	v plough tillage	increased the count of microorganisms in top 7 cm of soil		Experiment	7	United States				increase			Doran (1980)
No till	v plough tillage	reduced microorganism counts below the top 7 cm of soil		Experiment	7	United States				decrease			Doran (1980)
No till	v disc and chisel tillage	had no effect on fungi, bacteria levels in top 15 cm		Experiment	1	United States				similar			Mathew et al. (2012)
No till	v disc and chisel tillage	increased PLFA reading in top 15 cm		Experiment	1	United States				+65 nmol/g			Mathew et al. (2012)
No till	v intensive tillage	reduced machinery energy inputs		Article		United States						0.20-0.50	Huggins and Reganold (2008)
No till	v intensive tillage	reduced labour inputs		Article		United States					0.50-0.70		Huggins and Reganold (2008)

Colour code: Positive: Similar Negative: Inconclusive or confounding factors:

Table A.3. Evidence worksheet for cover cropping (CC)

Intervention (A)	Relative to baseline (B)	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/System B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (System A/System B)	GHG emission system A/System B	Water quality or nitrogen export	Runoff	Soil erosion: system A/System B	Weed incidence	Biodiversity	Costs	Reference
Non-leguminous CC	Bare fallow soil	similar yields		Meta-analysis	206	Global	Similar				0.30						Tonitto et al. (2006)
Leguminous CC	Bare fallow soil	similar yields		Meta-analysis	69	Global	Similar				0.60						Tonitto et al. (2006)
Cover cropping	Bare fallow soil	Increased soil carbon		Meta-analysis	139	Global		+0.32 Mg C ha/yr									Poeplau and Don (2015)
Leguminous CC	Bare fallow soil	Increased maize yield		Meta-analysis	80	USA & Canada	1.24										Miguez and Bollero (2005)
Grass CC	Bare fallow soil	No effect on maize yield		Meta-analysis	71	USA & Canada	0.99										Miguez and Bollero (2005)
Leguminous CC	Bare fallow soil	increased maize yield		Meta-analysis	65	USA & Canada	1.21										Marcillo and Miguez (2017)
Grass CC	Bare fallow soil	had no effect on maize yield		Meta-analysis	65	USA & Canada	1.00										Marcillo and Miguez (2017)
Orchard alley intercropping	Bare fallow soil	no impact on crop yields		Meta-analysis	11	Mediterranean	Similar	0.43-1.01 Mg ha^{-1} yr^{-1}	Increase								Morugán-Coronado et al. (2020)
Leguminous CC	Bare fallow soil	No effect on leaching		Meta-analysis	3	Global					1.00						Thapa et al. (2018)
Non-leguminous CC	Bare fallow soil	Reduction in leaching		Meta-analysis	27	Global					0.44						Thapa et al. (2018)
Non-leguminous CC	Bare fallow soil	Increases in soil C		Meta-analysis	144	Global		0.56 Mg C ha/yr	1.15								Jian et al. (2020)
Cover cropping	Bare fallow soil	No impact on CO2e		Farm study		Switzerland				Similar							Prechsl et al. (2017)
Cover cropping	Vineyard rows	Increases biodiversity		Farm study		Spain			1.29	1.01					1.38		Guzmán et al. (2019)
Cover cropping	Bare fallow soil	Reduce groundwater recharge		Meta-analysis	28	Global						Decrease recharge					Meyer et al. (2019)
Cover cropping	Bare fallow soil	resulted in similar yields in primary crop		Meta-analysis	106	Global	0.96-1.13										Abdalla et al. (2019)
Cover cropping	Bare fallow soil	Decreased N leaching & increases soil C		Meta-analysis	106	Global	0.96	0.05 Mg C ha/yr			Decrease	Decrease					Abdalla et al. (2019)
Cover cropping	Bare fallow soil	Reduce soil erosion		Lab study		UK							0.01 - 0.75				de Baets et al. (2011)
Cover cropping	Bare fallow soil	Decrease erosion		Review		Global			Increase				Decrease				Haruna et al. (2020)
Different cover crops	Bare fallow soil	affect mycorrhizae		Farm study		USA											Murrell et al. (2020)
Cover cropping	Bare fallow soil	Suppressed weeds		Meta-analysis	53	Global								0.05 - 0.6			Osipitan et al. (2019)
Cover cropping	Bare fallow soil	Increased costs and reduced gross margins		Review		UK										+£150/ha	AHDB (2020)
Cover cropping	Bare fallow soil	Farmer survey of UK cover-crop user experience		Farmer survey		UK										Increased labour costs	Storr et al. (2019)
Mixed species CC	Bare fallow soil	Increases fungi biomass		Incubation		Netherlands									Increase		Drost et al. (2020)
Cover cropping	Bare fallow soil	Decrease yields, increase biodiversity		Farm study		Italy	Reduced								Similar		Fiorini et al. (2022)

Table A.4. Evidence worksheet for organic crop production

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Inputs	Crop yield ratio: A/B	Additional carbon storage	Soil carbon (A/B)	GHG emission system A/System B	Pest or weed	Reference
Regenerative agriculture	Conventional			Desk study				*1.08*	*0.40-1.40*		*-0.23 t CO_2eq/ha/a*		*Drawdown Project (2017)*
Organic agriculture	Conventional	increased bean yield (organic had more irrigation)		LCA analysis	*2*	Greece		*1.12-1.32*					Abeliotis et al. (2013)
Organic maize/legume	Conventional maize/soya	resulted in similar (but less frequent) maize yields		Farm results	*1*	USA		1.00					Drinkwater et al. (1998)
Low input practices	Conventional	resulted in similar or lower yields		Article (no data)		USA		0.90-1.00					Kamenetzky and Maybury (1989)
Organic agriculture	Conventional	resulted in lower yields		LCA metaanalysis	37	Global		0.48-0.80					Clarke and Tilman (2017)
Organic agriculture	Conventional	resulted in lower yields		Meta-analysis	115	Global		0.81					Ponisio et al. (2015)
Organic agriculture	Conventional	resulted in lower yields		Meta-analysis	20	USA & Europe		0.74					Skinner et al. (2014)
Organic agriculture	Non-organic	resulted in lower yields		Meta-analysis	10	Developed		0.83					Mondelaers et al. (2009)
Organic farming	Conventional	reduced the yield of wheat and potatoes		Experimental	1	Germany		0.48-0.58					Lin and Hulsbergen (2017)
Organic horticulture	Non-organic	resulted in lower yields		Meta-analysis	300-560	Global		0.83					Lesur-Dumoulin et al. (2017)
Regen. Ag.	Conventional	resulted in lower maize yields		Field comparison	40 v 38	USA		0.71					LaCanne and Lundgren (2018)
Organic farming	Conventional	resulted in lower yields		Global	315	Global		0.75					Seufert et al. (2012)
Organic	Non-organic farming	reduced yields of wheat, barley, oats		Experiment plots	2	Norway		0.40-0.47					Korsaeth (2012)
Organic no-till	Organic-ploughing	resulted in lower yields		Meta-analysis	21	Europe		0.92					Cooper et al. (2016)
Organic no-till	Organic-ploughing	Increased weeds		Meta-analysis	21	Europe						1.56	
Adding organic inputs	Field with no nutrient def.	had statistically similar yields across most crops		Meta-analysis	107	Europe		1.01					Hijbeek et al. (2017)
Adding organic inputs	Field with no nutrient def.	resulted in higher yields with potatoes		Meta-analysis	11	Europe		1.07					Hijbeek et al. (2017)
Adding organic inputs	Field with no nutrient def.	resulted in higher yields with maize		Meta-analysis	15	Europe		1.04					Hijbeek et al. (2017)
Adding manure	Adding manure + P_2O_5	reduced crop yield		Experiment	1	Senegal		0.71					Diop AM (1999)
Adding manure	Not adding manure	increased sorghum yields		Meta-analysis	13	Africa		+480-880 kg/ha					Tonitto and Ricker-Gilbert (2016)
Not supplying N	Supplying synthetic N	reduced sorghum yields		Meta-analysis	13	Africa		-390-720 kg/ha					Tonitto and Ricker-Gilbert (2016)
Organic	Non-organic	increases organic matter inputs		Meta-analysis	71	Europe	1.35						Tuomisto et al. (2012)
Regenerative agriculture	Conventional	increases soil organic carbon		Field comparison	40 v 38	USA				1.09			LaCanne and Lundgren (2018)
Organic	Non-organic	increases soil organic matter		Meta-analysis	9	Developed				1.12			Mondelaers et al. (2009)
Organic	Non-organic	increases soil organic matter		Meta-analysis	71	Europe				1.07			Tuomisto et al. (2012)
Add organic amendments	no organic amendments	increases soil organic carbon		Meta-analysis	174	Mediterranean			+1.31 Mg/ha/yr				Aguilera et al. (2013)
Addition of manure	no addition of manure	increases soil organic carbon		Meta-analysis	298	Global			+1.8 g C/kg				Han et al. (2016)
Organic plough + legumes	Conventional	increased soil carbon		Field study	1	USA			+0.08 Mg/ha/yr				Robertson et al. (2000)
Organic cattle production	a maize rotation	increased soil carbon		Field study	1	USA			+0.1 Mg/ha/yr				Drinkwater et al. (1998)
Adding green manure	fallow in rotation	increased soil carbon storage		Meta-analysis	7	Canada			+150 kg C/ha/yr				VandenBygaart et al. (2003)
No chemical fertilizer	chemical fertilizer	decreases soil organic carbon		Meta-analysis	298	Global			-1.7 g C/kg				Han et al. (2016)
Organic apples	Conventional apples	had a higher level of soil carbon		Experiment	1					+1.17%			Kramer et al. (2006)
										1.08			
Regenerative agriculture	Conventional	reduced the numbers of a non-economic pest		Field comparison	40 v 38	USA						0.10	LaCanne and Lundgren (2018)
Organic agriculture	Conventional	increased GWP per unit food		LCA analysis	*2*	Greece					1.22-1.45		Abeliotis et al. (2013)
Organic agriculture	Conventional	increased GWP per unit area		LCA analysis	*2*	Greece					1.39-1.91		Abeliotis et al. (2013)
Addition of legumes	no legumes in rotation	reduced net GHG gas emissions		European model	1	Europe					-0.4 Mg CO_2e/ha/yr		Tuomisto et al. (2013)
Soil cover for whole year	incomplete soil cover	reduced net GHG gas emissions		European model	1	Europe					-0.3 Mg CO_2e/ha/yr		Tuomisto et al. (2013)
Organic agriculture	Conventional	reduced GWP per unit area		Meta-analysis	5	Developed					0.57		Mondelaers et al. (2009)
Organic juice production	Conventional	reduced GWP per unit food		LCA analysis	2	China & Brazil					0.60-0.85		Knudsen (2011)
Organic agriculture	Conventional	similar GWP per unit food		Meta-analysis	2	Developed					0.93		Mondelaers et al. (2009)
Organic agriculture	Conventional	similar global warming potential per unit food		LCA meta-analysis	37	Global					0.96		Clarke and Tilman (2017)
Organic agriculture	Conventional	reduced nitrous oxide emissions per unit area		Meta-analysis	20	Europe & USA					0.86		Skinner et al. (2014)
Organic agriculture	Conventional	reduced N2O emissions per area (less N applied)		Meta-analysis	10	Europe					0.69		Tuomisto et al. (2012)
Organic agriculture	Conventional	increased nitrous oxide emissions per unit food		Meta-analysis	20	Europe & USA					1.08		Skinner et al. (2014)
Organic agriculture	Conventional	used similar GHG gas emissions per unit food		Meta-analysis	23	Europe					1.00		Tuomisto et al. (2012)

Table A.4. Evidence worksheet for organic crop production (continued)

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Eutrophication or acidification potential	Water quality or nitrogen export	Biodiversity	Labour	Energy	Costs	Profit	Reference
Organic agriculture	Conventional	greater eutrophication potential per unit food		LCA meta-analysis	37	Global	1.37							Clarke and Tilman (2017)
Organic agriculture	Conventional	similar acidification potential per unit food		LCA meta-analysis	37	Global	0.87							Clarke and Tilman (2017)
Organic agriculture	Conventional	reduced nitrate leaching per area (less N applied)		Meta-analysis	71	Europe		0.69						Tuomisto et al. (2012)
Organic agriculture	Conventional	increased nitrate leaching per unit product		Meta-analysis	71	Europe		1.49						Tuomisto et al. (2012)
Organic apples + manure	Apples + synthetic fertiliser	reduced nitrate leaching (for constant N appliied)		Experiment	1			-1.1 mg NO_3-N						Kramer et al. (2006)
Organic agriculture	Conventional	had mixed effects on freshwater toxicity potential		LCA per m2/a	2	Greece								Abeliotis et al. (2013)
Organic agriculture	Conventional	reduced the less of nitrate leaching per unit area		Meta-analysis	14	Developed		0.68						Mondelaers et al. (2009)
Organic	Inorganic farming	resulted in depleted soil nitrogen and phosphorus		Experiment plots	2	Norway		-30 kg N - 8 kg P/ha/yr						Korsaeth (2012)
Organic agriculture	Conventional	has a higher energy output/energy input ratio		Review							consistent increase			Gomiero et al. (2011)
Organic agriculture	Conventional	uses less energy per unit food		LCA meta-analysis	37	Global					0.85			Clarke and Tilman (2017)
Organic agriculture	Conventional	uses less energy per unit food across all systems		Meta-analysis	37	Europe					0.79			Tuomisto et al. (2012)
Organic horticulture	Conventional	generally uses less energy per unit area		LCA study	1	UK					reduction			Metcalfe and McCormack (2000)
Organic horticulture	Conventional	generally uses less energy per unit food		LCA study	1	UK					reduction except carrots			Metcalfe and McCormack (2000)
Organic farming	Inorganic farming	reduces fossil-fuel based inputs		Review							0-50-0.70			Ziesmer (2007)
Organic farming	Non-organic farming	increases floral and faunal diversity		Review	21	Global			Consistent increase					Gomiero et al. (2011)
Organic farming	Conventional farming	increased arthropod abundance		Meta-analysis	81	Global			1.45					Lichtenberg et al. (2017)
Organic farming	Conventional farming	increased abundance of pollinator species		Meta-analysis	20	Global			1.90					Lichtenberg et al. (2017)
Organic farming	non-organic farming	increases biodiversity in most environments		Meta-analysis	396	Global			83% pos; 3% neg					Rahmann (2011)
Organic farming	Non-organic farming	increases species richness for most species groups		Meta-analysis	63	Global			1.30					Bengtsson et al. (2005)
Organic farming	Non-organic farming	increases the mean abundance of species		Meta-analysis	63	Global			1.50					Bengtsson et al. (2005)
Organic farming	Conventional farming	increased arthropod abundance		Meta-analysis	81	Global			1.10-1.21					Lichtenberg et al. (2017)
Organic farming	Conventional farming	increased abundance of pollinator species		Meta-analysis	20	Global			1.32-1.55					Lichtenberg et al. (2017)
Organic farming	Non-organic farming	increases the mean abundance of weed species		Meta-analysis	5	Global			1.50					Bengtsson et al. (2005)
Organic farming	Non-organic farming	did not significant affect the species richness of soil organisms		Meta-analysis	63	Global			Positive but not significant					Bengtsson et al. (2005)
Low input farming	Conventional farming	resulted in higher species richness		Modeling		Global			1.64					Elshout et al. (2014)
Adding manure	Not adding manure	increased crop revenue from sorghum		Meta-analysis	13	Africa							+$133-176/ha	Tonitto and Ricker-Gilbert (2016)
Regenerative agriculture	Conventional	resulted in lower cost of production		Field comparison	40 v 38 field	USA						0.58		LaCanne, CE, Lundgren JG (2018)
No fertiliser input	conventional beef	reduced net returns		Article (no data)		USA							Reduced	Kamenetzky and Maybury (1989)
Organic maize/ legume rotation	Conventional Maize/soya rotation	requires more labour		Farm comparison	1	USA				Higher				Hanson et al. (1997)
Organic farming	Inorganic farming	increases labour requirements		Review						Higher				Gomiero et al. (2011)
Organic farming	Inorganic farming	increases labour requirements		Review						1.30-1.35				Ziesmer (2007)
Organic farming	Inorganic farming	increases labour costs		Meta-analysis	129	Global				1.07-1.13				Crowder and Reganold (2015)
Organic farming	Inorganic farming	reduces profitability (if no organic premium)		Meta-analysis	129	Global							0.73-0.77	Crowder and Reganold (2015)
Organic farming	Inorganic farming	increases profitability (with organic premium)		Meta-analysis	129	Global							1.22-1.35	Crowder and Reganold (2015)

Table A.5. Evidence worksheet for integrated crop livestock systems (ICLS) relative to a) crop systems and b) livestock systems, and for integrated pest management (IPM)

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/System B	Soil carbon (System A/System B)	GHG emission system A/System B	Leaching	Pest or weed numbers	Agrochemical use	Biodiversity	Variability of profit	Net margin	Reference
ICLS	Crop system	Increase soil carbon		Experiment	1	Brazil		1.20								Carvalho et al. (2010)
ICLS	Continuous maize	Increased mean maize yields		Experiment	1	USA	1.06									Maughan et al. (2009)
ICLS	Cropland	Increased yields; no effect on CO2 efflux		Field Study	1	USA	1.09									Tracy et al. (2008)
ICLS	Cropland	Soil carbon similar after 4 years		Field study	1	USA		Similar								Tracy et al. (2008)
ICLS	Cropland	Increased abundance of bees		Field study	1	Canada							3.52			Morandin et al. (2007)
ICLS	Cropland	Increased farmer profit		Review		Global								Decreased		Sekaran et al. (2021)
ICLS	Cropland	Reduced herbicides; increases in soil C		Review		Global		Increase			Decrease					Peyraud et al. (2014)
ICLS	Cropland	Increased GWP		Field Study	1	USA			4.73							Liebig et al. (2021)
ICLS	Cropland	Comparable yields		Meta-analysis	66	Global	0.93-1.02									Peterson et al. (2020)
ICLS	Cropland	Tended to increase soil carbon		Experiment	1	Brazil		1.09								Salton et al. (2014)
ICLS	Cropland	Tended to reduce nutrient leaching		Review		USA				Reduced						Sanderson et al. (2013)
ICLS	Cropland	Enhanced soil carbon sequestration		Review	3	USA		Increase								Hilimire (2011)
ICLS	Cropland	Reduced interannual variability in gross margin		Review	1	Australia								0.68		Bell and Moore (2012)
ICLS	Cropland	Increased arthropod biodiversity		Field Study		Germany							Increase			Tamburini et al. (2022)
ICLS	Cropland	Increased yields		Review		Global					Decrease		Increase			Garrett et al. (2017)
Add cereal	Grass system	Positive effect on winter forage availability		Experiment	1	Australia	2.82									Bell et al. (2015)
Add cereal	Grass system	Positive effect on sheep grazing days		Experiment	1	Australia	1.12									Dove et al. (2015)
ICLS	Permanent pasture	Tended to reduce soil carbon		Experiment	1	Brazil		0.96								Salton et al. (2014)
ICLS	Pasture	No effect on soil carbon		Experiment	1	Brazil		No change								de Sant-Anna et al. (2017)
ICLS	Pasture	Reduced interannual variability in gross margin		Review	1	Australia								0.68		Bell and Moore (2012)
ICLS	Pasture	Reduce SOC		Review	1	UK		Reduce								Powlson et al. (2011)
ICLS	Pasture	should increase plant biodiversity		Model	1	UK							Increase			White et al. (2019)
IPM	Baseline practice	Increased yields		Review	61	USA	1.14									Norton and Mullen (1994)
IPM	Baseline practice	Reduced agrochemical use		Review	61	USA						0.85				Norton and Mullen (1994)
IPM	Baseline practice	Increased net margins		Review	61	USA									1.48	Norton and Mullen (1994)

Table A.6. Evidence worksheet for field margins and agri-environment schemes (AES)

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/System B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (System A/System B)	Soil erosion: system A/System B	Pest or weed numbers	Biodiversity	Reference
AES		Positive plant biodiversity		Meta-analysis	14	Europe						6 +; 7 -/+; 2 -	Kleijn and Sutherland (2003)
AES		Increase arthropod diversity		Meta-analysis	17	Europe						11+;, 3 +/-; 3 0	Kleijn and Sutherland (2003)
AES		Positive effect on bird diversity		Meta-analysis	19	Europe						4+; 9+/-; 2-	Kleijn and Sutherland (2003)
Field with 15% wild flower meadow	Conventional farming	Increased abundance of bees (1 ha) field		Field-study	10	Germany	0.85					7.1	Batáry and Tscharntke (2022)
Farm with 5% wild flower meadow	Conventional farming	Increased abundance of bees (100 ha)		Field-study	10	Germany	0.95					2.7	Batáry and Tscharntke (2022)
Planting of wildflowers near Blueberries	No wildflowers	Increased yield of blueberries		Field-study	1	USA	1.2						Garibaldi et al. (2014)
Arable land	Grass margin with hedge on 5% of land	Increase annual soil carbon sequestration		Model		UK	0.95	0.045					Falloon et al. (2004)
Arable land	Grass margin with hedge on 5% of land	Increased biomass carbon storage		Model		UK	0.95	2.7					Falloon et al. (2004)
Yields at edge of field (0-9 m)	Rest of field	Reduced wheat, bean and OSR yields		Field study	3	UK	0.75						Pywell et al. (2015)
Creation of habitats on 8% of land	Conventional farming	Increased mean yields/ha of beans		Field study	3	UK	1.24						Pywell et al. (2015)
Creation of habitats on 8% of land	Conventional farming	Similar yields/ha of OSR and wheat		Field study	3	UK	0.92						Pywell et al. (2015)
Hedgerows	Middle of field	Tend to decrease crop yields		Review		UK	Reduce						Marshall and Moonen (2002)
Hedgerows	Agricultural landscape	Increases soil carbon stock		Field-study		France			1.25				Follain et al. (2007)
Hedgerows	Cultivated field	Increased soil carbon		Field study		USA			1.36				Chiartas et al. (2022)
Hedgerows	arable control	Increased soil carbon		Meta-analysis	38	Global temperate			1.32				Drexler et al. (2021)
Hedgerows	grassland	similar soil carbon		Meta-analysis	45	Global temperate			0.91				Drexler et al. (2021)
Hedgerows	Arable field	Increased hydraulic conductivity; hence less runoff		Field study		UK							Holden et al. (2019)
6 m grass strips	No grass strips	Reduced sediment losses		Field study	3	France				0.04			Patty et al. (1997)
Field margins	Middle of field	Can be a source of weeds		Review		Europe							Marshall (2005)
AES (Winter stubbles)	Historic land use	Increase in population of Cirl Bunting (1993-1998)		Field-study	1	Devon, UK						1.3	Wooton et al. (2000)
AES (including organic) on cropland in simple landscapes	Areas without AES	Increased species richness of arthropods		Meta-analysis	31	Global						1.8	Batáry et al. (2011)
AES (including organic) on cropland in complex landscapes	Areas without agri-environment scheme	No effect on species richness of arthropods		Meta-analysis	8	Global						0.8	Batáry et al. (2011)
Agri-environment schemes (including organic) on grassland	Areas without agri-environment scheme	Positive effect on arthropod species		Meta-analysis	38	Global						1.6	Batáry et al. (2011)
Areas with off-field agri-environment scheme	Areas without off-field agri-environment scheme	Increased species diversity		Meta-analysis	35	Europe						2.6	Batáry et al. (2015)
Site close to hedgerow	Site away from hedgerow	Increased presence of bees and hoverflies		Field-study	4	UK						Increased	Marini et al. (2016)
Hedgerows	Field without hedgerow	Increased presence of mollusc pests		Review		UK					Increased		Marshall (2005)
Field Margin	Centre of cereal field	Increased presence of artropod predators		Field-study	1	Norway						5.0	Dennis and Fry (1992)
Hedgerows	Arable field	Increased earthworm density		Field study		UK						2.1	Holden et al. (2019)
Hedgerows	Pasture field	Had no significant effect on earthworm density		Field study		UK						0.9	Holden et al. (2019)
Set-aside	Arable field	Increased plant species richness		Field study		Hungary						2.5	Kovács-Hostyánszki et al. (2011)
Set-aside	Arable field	Increased species richness of butterflies		Field study		Hungary						4.0	Kovács-Hostyánszki et al. (2011)
Set-aside	Semi-natural grassland	Did not affect plant species richness		Field study		Hungary						1.0	Kovács-Hostyánszki et al. (2011)
Set-aside	Semi-natural grassland	Similar species richness of butterflies		Field study		Hungary						1.0	Kovács-Hostyánszki et al. (2011)
New wildflower strip	grassland	Increased pollinator abundance		Field study		Germany						3.6	Krimmer et al. (2019)

Table A.7. Evidence worksheet for pasture-fed livestock production

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Inputs	Crop yield ratio: system A/System B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (System A/System B)	GHG emission system A/System B	Eutrophication or acidification potential	Biodiversity	Costs	Reference
Pasture-fed livestock (20 months)	Pasture + concentrate (20 months)	Reduced meat production		LCA study		Ireland		0.68							Herron et al. (2021)
Diverse pasture	Ryegrass-clover pasture	Increased soil C sequestration		Field study		New Zealand			1.2						McNally et al. (2015)
PLFA farms	Non PLFA farms	Positive effect on plant species richness		Field study		UK				No change			Increase		Norton et al. (2022)
Diverse pasture (8 species)	Ryegrass monoculture	Biodiversity increased soil carbon sequestration		Field study		Netherlands				1.17					Cong et al. (2014)
Pasture-fed livestock	Grain-fed livestock	Increased land requirement and carbon footprint		LCA study		North America					1.42				Capper (2012)
Pasture-fed livestock	Grain-fed livestock	Increased GHG emissions and land use requirements		Meta-analysis	7	Global	Increased				1.19				Clark and Tilman (2017)
Pasture-fed livestock	Grain-fed livestock	Increased nutritional security		Review		Global						Decrease			Smith et al. (2013a)
Pasture-fed livestock	Grain-fed livestock	Decreased costs for increasing grain feed		Field study		Europe								0.67	Dillon et al. (2008)
Pasture-fed livestock	Grain-fed livestock	Perceived human health and animal welfare benefits		Review		Global									Stampa et al. (2020)

Table A.8. Evidence worksheet for multi-paddock grazing

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/system B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (A/B)	Water quality	Runoff	Soil erosion: A/B	Costs	Profit	Reference
Managed grazing	*conventional grazing*			*Desk study*		*Global*	*1.10*	*0.63*						*1.74*	*Drawdown (2017)*
Multi-paddock	Continuous grazing	resulted in increased stocking rates		Experiment	1	Australia	1.07-1.22								Badgery et al. (2017)
Multi-paddock	Continuous grazing	used higher stocking rates		Experiment	1	Texas USA									Heitschmidt et al. (1982)
Multi-paddock	Continuous grazing	resulted in greater grass consumption		Modelled	1	USA	Generally positive but dependent on rotation length and stocking density								Chen and Shi (2018)
Multipaddock	Continuous grazing	increases consumption of palatable grasses		Modelled		USA	1.09								Wang et al. (2016)
Multipaddock	Continuous grazing	resulted in similar pasture productivity		Experimental	12 v 11	South Australia	about 1								Sanderman et al. (2015)
Multi-paddock	Continuous grazing	resulted in similar grass yields		Meta-analysis	75	Global	1.00								Hawkins (2017)
Multipaddock	Continuous grazing	results in similar yields		Experimental	9 years	Central Plains, USA	0.98						0.993		Derner and Hart (2007)
Multi-paddock	Continuous grazing	resulted in similar liveweight gains per hectare		Meta-analysis	75	Global	+7 kg/ha/d								Hawkins (2017)
Multi-paddock	Continuous grazing	resulted in reduced herbage quality		Experiment (3.5 ha plots)	1	Australia									Cox et al. (2017)
Multipaddock	Continuous grazing	plots had a higher soil organic matter concentration		Experimental	1	USA			1.15						Teague et al. (2010)
Multipaddock	Continuous grazing	plots had a higher soil organic matter concentration		Experimental	1	USA			1.50						Teague et al. (2011)
Multipaddock	Continuous grazing	resulted in similar soil organic matter levels		Experimental	12 v 11	South Australia	1		0.99						Sanderman et al. (2015)
Multipaddock	Continuous grazing	increased perennial grass cover		Pairwise comparison	2	Botswana	+20%								Mudongo et al. (2016)
Multipaddock	Continuous grazing	decreased tree cover		Pairwise comparison	2	Botswana	-7 to -17%								Mudongo et al. (2016)
Multipaddock	Continuous grazing	decreased surface runoff		Modelled (with experimental data)	4x study ranches	Texas, USA					0.53				Park et al. (2017)
Multipaddock	Continuous grazing	increased infiltration rates		Experimental	1	USA					1.34				Teague et al. (2010)
Multipaddock	Continuous grazing	increased soil aggregate stability		Experimental	1	USA				1.15					Teague et al. (2011)
Multipaddock	Continuous grazing	decreased sediment loss		Experimental	1	USA						0.22			Teague et al. (2011)
Multi-paddock	Continuous grazing	resulted in increased management costs		Meta-analysis observation	75	Global							Increased costs		Hawkins (2017)

Table A.9. Evidence worksheet for organic livestock systems

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio A/B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (A/B)	GHG emission system A/B	Water quality	Biodiversity	Energy	Reference
Organic grassland														
Grass receiving FYM	Grass receiving NPK	increased grass yield		Field comparison	1	England	1.50							Kidd et al. (2017)
Organic grass (+ 125 kg N/ha from legumes)	Grass receiving 125 kg N/ha	increased grass yield		Field comparison	1	Scotland	1.22							Topp et al. (2007)
Adding organic amendments	Not adding amendments	increased dry matter production on rangelands		Meta-analysis	92	Global	1.98							Table S2 (Gravuer et al. 2019).
Organic dairy	Conventional dairy	reduced milk yield per cow		Farm comparison	15	Sweden	0.93							Mueller et al. (2014)
Organic dairy	Conventional dairy	reduced milk yield per agricultural area		Farm comparison	15	Sweden	0.70							Mueller et al. (2014)
Grass-fed beef	Grain-fed beef	has lower output per unit land		LCA meta-analysis	4	Global	0.71							Clarke and Tilman (2017)
Grass-fed beef	Grain-fed beef	similar GHG emissions per unit food		LCA meta-analysis	7	Global				1.19				Clarke and Tilman (2017)
Organic diary	Non-organic dairy	increased GHG emissions per unit milk		Review	3	Global				1.13				Gomiero et al. (2011)
Grass receiving FYM	Grass receiving NPK	increased soil carbon		Field comparison	1	England			1.20					Kidd et al. (2017)
Adding organic amendments	Not adding amendments	increased soil carbon levels on rangelands		Meta-analysis	92	Global			1.30					Table S2 (Gravuer et al. 2019).
Addition of legumes	before legumes	increased soil carbon sequestration		Review	6	Global		+0.75 Mg C/ha/yr						Conant et al. (2001)
Addition of earthworms	before earthworms	increased soil carbon sequestration		Review	2	Global		+2.35 Mg C/ha/yr						Conant et al. (2001)
Adding organic amendments	Not adding amendments	reduced runoff from rangelands		Meta-analysis	92	Global					0.49			Table S2 (Gravuer et al. 2019).
Adding organic amendments	Not adding amendments	increased the concentration of nitrate in runoff		Meta-analysis	92	Global					5.59 for N 8.96 for P			Table S2 (Gravuer et al. 2019).
Adding organic amendments	Not adding amendments	had no statistical effect on native plant communities		Meta-analysis	92	Global						0.94		Table S2 (Gravuer et al. 2019).
Organic dairy production	Conventional dairy	reduced the biodiversity damage impact		Modelling study	1	Sweden						0.42		Mueller et al. (2014)
Organic grass (receiving 125 kg N/ha from legumes)	Grass receiving 125 kg N/ha	increased energy efficiency (energy out/energy in)		Field comparison	1	Scotland							3.02	Topp et al. (2007)
Organic dairy	Conventional dairy	generally reduced energy use per litre of milk		Review	7	Global							0.78	Gomiero et al. (2011)
Organic dairy	Conventional dairy	reduced energy use per hectare		Farm study	1	Denmark							0.67	Dalgaard (2013)
Organic dairy	Conventional dairy	reduced energy use per cow		Farm study	1	Denmark							0.77	Dalgaard (2013)

Table A.10. Evidence worksheet for tree crops

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Inputs	Crop yield ratio: system A/System B	Additional carbon storage	Soil carbon (System A/System B)	GHG emission system A/System B	Biodiversity	Profit	Reference
Tree crops														
Tropical staple trees	*Annual crops on degraded land*			*Desk study*	*9*	*Global*		*2.40*	*4.70* $t\ C\ ha^{-1}\ a^{-1}$					*The Drawdown project (2017) on degraded land*
Plantation	Cropland	increased soil carbon		Meta-analysis	74	Global				1.18				Guo and Gifford (2002)
Shaded perennial system	Agriculture	increased soil carbon		Review/meta-analysis	2	Global				1.01				Kim et al. (2016)
Bananas	Maize	increased calorie production		Model	1	Rwanda		1.60						Bidogeza et al. (2015)
Bananas	Maize	reduced protein production		Model	1	Rwanda		0.75						Bidogeza et al. (2015)
Agroforestry	Degraded arable and grassland	can increase above ground carbon sequestration		Review		Global			0.4-2.8 t C/ha/yr	0.2-0.6 t C/ha/yr				Mutuo et al. (2005)
Fruit trees	Arable	increased the potential carbon sequestration by plants		Regional study	2	Bari				2-28 t CO_2/ha/yr				Dal Sasso et al. (2012)
Tree plantation	Agricultural land	had no significant effect on nitrous oxide emissions		Review/meta-analysis	1						-1.4 kg NO_2/ha/yr			Kim et al. (2016)
Orchard	Arable cropping	Increases biodiversity of arthropods and insectivorous birds		Review	1	Global						increases		Simon et al. (2010)

Table A.11. Evidence worksheet for tree intercropping

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio: A/B	Additional carbon storage	Soil carbon (A/B)	GHG emission system A/System B	Water quality or nitrogen export	Soil erosion: A/ B	Biodiversit y	Labour	Energy	Profit	Reference
Tree-intercropping	*Annual crops*			*Desk study*		*Global*		*0.90-2.70*								*1.02*	*The Drawdown project (2017) on degraded land*
Tree intecropping with soybean	Soybean production	increased potential carbon sequestration		Field experiment	1	Canada		+0.84 to +2.12 relative to -1.15 tC/ha/yr									Thevathasan et al. (2016)
Silvoarable agroforestry	Arable	increased carbon sequestration		Modeling	1	UK		+4 t CO2/ha/yr									Garcia de Jalón et al. (2018b)
Silvoarable agroforestry	Arable	increased carbon sequestration		Review		Europe		+2.75 tC/ha/yr									Aertsens et al. (2013)
Intercropping	Arable	increases soil organic content		Review/meta-analysis	4	Global			1.16								Kim et al. (2016)
Silvoarable	Arable	Increases biodiversity and wildlife habitat		Interviews	58	Europe							Increases				Garcia de Jalón et al. (2018a)
Silvoarable agroforestry	Arable	increased biodiversity		Meta-analysis		Europe							1.37				Torralba et al. (2016)
Silvoarable agroforestry	Arable	reduced food production		Experiment and model	1	UK	0.42										Garcia de Jalón et al. (2018b)
Silvoarable agroforestry	Arable	maintained food production		Experiment	1	Germany	0.95										Kanzler et al. (2018)
Adding hedges and landscape features	arable landscape	reduced net GHG gas emissions in JRC model		European model	1	Europe				-0.1 Mg CO_2e/ha/yr							Tuomisto et al. (2013)
Silvoarable agroforestry	Arable	reduced CO_2 emissions		Modeling	1	UK				0.46							Garcia de Jalón et al. (2018b)
Intercropping	Arable	increased NO_2 emissions		Meta-analysis	4	Global				+1.0 kg NO_2/ha/yr							Kim et al. (2016)
Silvoarable agroforestry	Arable	reduced soil erosion losses		Modeling	1	UK						0.50					Garcia de Jalón et al. (2018b)
Silvoarable agroforestry	Arable	reduced nitrogen surplus		Modeling	1	UK					-22 kg N/ha/yr						Garcia de Jalón et al. (2018b)
Silvoarable agroforestry	Arable	reduced erosion losses		Meta-analysis		Europe						0.40					Torralba et al. (2016)
Increasing tree cover	no increase in tree cover	reduced sediment loss in an extreme rainfall year		Watershed review		Iowa						0.05					Asbjornsen et al. (2013)
Increasing tree cover	no increase in tree cover	reduced nitrogen export in an extreme rainfall year		Watershed review		Iowa					0.15						Asbjornsen et al. (2013)
Silvoarable agroforestry	Arable farm	increased the energy produced per unit energy input		Experimental farm	1	Germany									1.18		Lin et al. (2017)
Silvoarable	Arable	increases management costs and labour		Interviews	58	Europe								increased			Garcia de Jalón et al. (2018a)
Silvoarable agroforestry	Arable	increased and reduced net margins		Modeling	42	Europe										Some positive; some negative	Graves et al. (2007)
Silvoarable agroforestry	Arable	reduced net margin		Modeling	1	UK										-€196/ha/yr	Garcia de Jalón et al. (2018b)
Silvoarable agroforestry	Arable	similar societal benefits		Modeling	1	UK										1.10	Garcia de Jalón et al. (2018b)

Table A.12. Evidence worksheet for multistrata agroforestry

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Inputs	Crop yield ratio: system A/System B	Additional carbon storage	Soil carbon (System A/System B)	GHG emission system A/System B	Biodiversity	Profit	Reference
Multistrata agroforestry														
Multistrata agroforestry	*Degraded grassland*			*Desk study*		*Global*		*NA*	*7.00* t C ha^{-1} a^{-1}				*NA*	*The Drawdown project (2017) on degraded land*
Fruit trees	Arable	increased the potential carbon sequestration by plants		Regional study	2	Bari				2-28 t CO_2/ha/yr				Dal Sasso et al. (2012)
Banana/coffee	Banana	increased soil carbon		Survey	1	Uganda				1.57				Zake et al. (2015)
Shaded perennial system	Agriculture	increased soil carbon		Review/meta-analysis	2	Global				1.01				Kim et al. (2016)
Plantation	Cropland	increased soil carbon		Meta-analysis	74	Global				1.18				Guo and Gifford (2002)
Cocoa and coffee agroforestry	Cocoa and coffee plantation	increased biodiversity		Meta-analysis	74	Global						Positive		De Beenhouwer et al. (2013)
Complex agroforestry	Simple agroforestry	Increased biodiversity		Meta-analysis	44	Brazil						1.15		Santos et al. (2019)
Agroforestry Cocoa	Conventional cocoa	resulted in reduced cocoa yields		Experiment	1	Bolivia		Cocoa production decreased						Niether et al. (2019)
Agroforestry Cocoa	Conventional cocoa	resulted in similar total crop yields		Experiment	1	Bolivia		Total crop production maintained						Niether et al. (2019)
Agroforestry Cocoa	Conventional cocoa	resulted in increased cocoa yields		LCA	60 farms	Colombia		3.00						Ortiz-Rodriguez et al. (2016)
Agroforestry Cocoa	Conventional cocoa	increased above ground carbon storage		Experiment	1	Bolivia			4.00 ratio					Niether et al. (2019)
Shaded perennial system	Agriculture	had no significant effect on nitrous oxide emissions		Review/meta-analysis	5	Global					+5.5 kg NO_2/ha/yr			Kim et al. (2016)

Table A.13. Evidence worksheet for silvopasture systems

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio: system A/System B	Additional carbon storage	Soil carbon (A/B)	GHG emission system A/System B	Soil erosion A/B	Biodiversity	Labour	Profit	Reference
Silvopasture	Business as usual grazing			*Desk study*	*"4-8 sources"*	*Global*	*1.10*	*4.80*						*3.79*	*Drawdown Project (2017)*
Silvopasture	Pasture	resulted in a similar level of food production		Meta-analysis	*82*	Europe	1.18								Torralba et al. (2016)
Silvopasture	Pasture	reduced the herbage yield		Field measurements	1	Italy	0.77								Seddaiu et al. (2018)
Silvopasture	Pasture	reduced herbage yield where grass was fertilised		Survey	1	Spain	reduced								Moreno et al. (2007)
Silvopasture	Pasture	increased herbage yield where grass was not fertilised		Survey	1	Spain	increased								Moreno et al. (2007)
Silvopasture	Pasture	enhances animal health and welfare		Interviews	187	Europe	Enhances animal health and welfare								Garcia de Jalón et al. (2018a)
Silvopasture	Pasture	increases carbon storage		Review	1	Europe		2 t C/ha/yr							Aertsens et al. (2013)
Silvopasture	Pasture	increases soil carbon storage		Field measurements	1	Italy			1.18						Seddaiu et al. (2018)
Silvopasture	Pasture	similar soil carbon storage		Field measurements	1	UK			1.00						Upson et al. (2016)
Silvopasture	Pasture	increases soil carbon at 0-15 cm		Meta-analysis	2	Global			1.05						De Stefano and Jacobson (2018)
Silvopasture	Pasture	enhances soil fertility		Meta-analysis	*82*	Europe			1.07						Torralba et al. (2016)
Integration of crops, trees and livestock	Conventional agriculture	reduced net GHG gas emissions		LCA		Brazil				0.45					Costa et al. 2018
Silvopasture	Pasture	enhances erosion control		Meta-analysis	*82*	Europe					0.37				Torralba et al. (2016)
Silvopasture	Pasture	enhances biodiversity		Interviews	187	Europe						Enhances			Garcia de Jalón et al. (2018a)
Silvopasture	Pasture	enhances biodiversity		Meta-analysis	*82*	Europe						1.21			Torralba et al. (2016)
Silvopasture	Pasture	enhances gamma biodiversity		Field measurements	1	Italy						1.31			Seddaiu et al. (2018)
Silvopasture	Pasture	increases labour and management costs		Interviews	187	Europe							Increases		Garcia de Jalón et al. (2018a)

Table A.14. Evidence worksheet for rewilding and agricultural land abandonment

Intervention A	Baseline B	Impact		Type of study	Number of studies	Location	Crop yield ratio A/B	Additional carbon storage (t C ha^{-1} a^{-1})	Soil carbon (A/B)	GHG emission system A/B	Water quality	Biodiversity	Energy	Reference
Land abandonment and rewilding														
Land abandonment	Agricultural land	reduces food production		Case study	1	Italy	Decreased							Smiraglia et al. (2016)
Annual sale of 75 t high value beef, pork and venison from rewilding project across 1100 ha	Mean UK lowland lamb production/ha across 1100 ha	reduces quantity of meat production		Case study	1	UK	0.11 = 75 t /660 t							Spencer (2017); Redman (2018)
Rewilded land	Agricultural land	reduces grass and crop production		Case study	1	Spain	0.80							Cerqueira et al. (2012)
Tropical reforestation	Agricultural land	above ground regrowth during first 20 years		Meta-analysis	143	Tropics		+6.4 Mg/ha/yr						Silver et al. (2000)
Perennial vegetation	degraded agricultural land	increases soil carbon (over 100 cm)		Review	11	USA		+0-660 kg C/ha/yr						McLauchlan (2006)
Pasture	Cultivation	increased soil carbon sequestration		Review	23	Global		+1.01 Mg C/ha/yr						Conant et al. (2001)
Pasture	Cropland	increased soil carbon		Meta-analysis	74	Global			1.19					Guo and Gifford (2002)
Native soil	Agricultural land	decreased soil carbon		Meta-analysis	50	Canada			1.32					VandenBygaart et al. (2003)
Abandonment	Extensive grazing	Reduced biodiversity		Review		Global						Decreased		Rey Benayas et al. (2007)
Abandonment	Agricultural land	increased short-term biodiversity		Review		Global						Increased		Lasanta et al. (2015)
Abandonment	Agricultural land	increased mega fauna abundance		Review		Global						Increased		Ceaușu et al. (2015)
Rewilding	Agricultural land	Increase invasive species		Review		Global						Increased invasives		Corlett (2016)