LUPG

The UK statutory conservation, countryside and environment agencies

The Role of Agroecology in Sustainable Intensification

Lampkin, N.H., Pearce, B.D., Leake, A.R., Creissen, H., Gerrard, C.L., Girling, R., Lloyd, S., Padel, S., Smith, J., Smith, L.G., Vieweger, A., Wolfe, M.S.

The Organic Research Centre Elm Farm

Game & Wildlife Conservation Trust

June 2015

Natural England

Cyfoeth Naturiol Cymru Natural Resources Wales

Scottish Natural Heritage All of nature for all of Scotland

SEPA Scottish Environment Protection Agency

NIEA Northern Ireland Environment Agency www.ni-environment.gov.uk

Environment Agency

Disclaimer

This report was produced by the authors on behalf of the Land Use Policy Group (LUPG). The views expressed within the report are those of the contractors and do not necessarily reflect the views of the agencies within LUPG.

This report should be quoted as:

Lampkin, N.H., Pearce, B.D., Leake, A.R., Creissen, H., Gerrard, C.L., Girling, R., Lloyd, S., Padel, S., Smith, J., Smith, L.G., Vieweger, A., Wolfe, M.S., 2015. The role of agroecology in sustainable intensification. Report for the Land Use Policy Group. Organic Research Centre, Elm Farm and Game & Wildlife Conservation Trust.

Corresponding author: Prof Nic Lampkin (nic.l@organicresearchcentre.com)

Copyright

For further information, the LUPG contact is:

Cécile Smith, Scottish Natural Heritage (cecile.smith@snh.gov.uk)

LUPG

LUPG comprises Natural England, Natural Resources Wales, Scottish Natural Heritage, the Environment Agency, Northern Ireland Environment Agency, Scottish Environment Protection Agency. LUPG provides independent evidence and analysis to Government on matters of common concern related to agriculture, woodlands and other rural land uses. It seeks to develop a common understanding of the pros and cons of policy mechanisms related to land use, particularly farming and forestry.

Natural England

Natural England is the government's advisor on the natural environment. We provide practical advice, grounded in science, on how best to safeguard England's natural wealth for the benefit of everyone. Our remit is to ensure sustainable stewardship of the land and sea so that people and nature can thrive. It is our responsibility to see that England's rich natural environment can adapt and survive intact for future generations to enjoy.

www.naturalengland.org.uk

Natural Resources Wales

Natural Resources Wales is a Welsh Government Sponsored Body. Our Purpose is to ensure that the environment and natural resources of Wales are sustainably maintained, sustainably enhanced and sustainably used, now and in the future.

http://naturalresources.wales/

Scottish Natural Heritage

Scottish Natural Heritage (SNH) is a government body established to secure conservation and enhancement of Scotland's unique and valued natural heritage – the wildlife, habitats and landscapes that have evolved in Scotland through long partnership between people and nature. SNH advises on policies and promotes projects that aim to improve the natural heritage and support its sustainable use. Its aim is to help people to enjoy Scotland's natural heritage responsibly, understand it more fully and use it wisely so it can be sustained for future generations.

www.snh.org.uk

The Environment Agency

The Environment Agency (EA) is the leading public organisation for protecting and improving the environment in England and Wales. The EA achieves this by regulating industry, waste and water quality; managing flood risk and water resources, and improving wildlife habitats in addition to many other activities. The EA also monitors the environment, and makes the information that it collects widely available.

www.environment-agency.gov.uk

Northern Ireland Environment Agency

The Northern Ireland Environment Agency takes the lead in advising on, and in implementing, the Government's environmental policy and strategy in Northern Ireland. The Agency carries out a range of activities, which promote the Government's key themes of sustainable development, biodiversity and climate change. Our overall aims are to protect and conserve Northern Ireland's natural heritage and built environment, to control pollution and to promote the wider appreciation of the environment and best environmental practices.

www.ni-environment.gov.uk

The Scottish Environment Protection Agency

The Scottish Environment Protection Agency (SEPA) is Scotland's environmental regulator. SEPA's main role is to protect and improve the environment, and it does so by regulating activities that can cause pollution, and by monitoring the quality of Scotland's air, land and water. SEPA reports on the state of Scotland's environment and publishes a wide range of environmental data and information.

http://www.sepa.org.uk/

FOREWORD

The capacity of the global food system to support a rising world population while preserving healthy ecosystems is the subject of much debate. Going back to 2007-08, the global spike in commodity prices highlighted that demand for food was starting to rise faster than supply. A range of factors are responsible for this trend. Failures in distribution, wastage along supply chains and inequalities in purchasing power are among the structural problems affecting food availability. The ongoing rise in the global population and the expansion of the middle-class within many countries is affecting the magnitude and the nature of the demand for food, with a shift to diets which are richer in animal-proteins. The ongoing rise in global temperatures, increasingly changeable weather patterns and greater competition for land, energy and water will affect the global food system as well as the ecosystem services which underpin agriculture and the natural environment in general[1].

In 2009, The Royal Society addressed the challenge of how food availability might be increased without repeating the environmental damage of the mid-20th Century - and discussed the concept of ‘sustainable intensification’ of global agriculture in which yields are increased without adverse environmental impact and without the cultivation of more land. This concept was developed in more detail in the Foresight report on the Future of Food and Farming, which described sustainable intensification as *“simultaneously raising yields, increasing the efficiency with which inputs are being used and reducing the negative environmental effects of food production*”[2].

Working through the Land Use Policy Group (LUPG), the statutory conservation, countryside and environmental agencies from across the UK are able to collaborate on a wide range of issues relating to land management. As such we have been engaging with the concept of sustainable intensification for a number of years. For example, a previous LUPG report considered how the sustainable intensification concept could be applied at the level of individual farms. In particular, the project aimed to explore whether there were examples of farmers increasing yields at the same time as reducing negative environmental impacts – or even enhancing the environment on their farms. The resulting piece of work - *“Exploring the Concept of Sustainable Intensification”* - was undertaken by John Elliott of ADAS and Professor Les Firbank of Leeds University and published in January 2013. The final report showed that out of a sample of twenty cutting-edge farms across Great Britain, four of these appeared to be carrying out sustainable intensification according to the research methodology. The project also demonstrated the need for a range of mutually agreed indicators and metrics that can be used to assess whether or not individual farms are on a path towards sustainable intensification.

A significant amount of work is currently taking place under the auspices of Defra’s Sustainable Intensification Platform. The LUPG agencies are keen to avoid duplication and fund research work only where we can add value.

---

[1] International Assessment of Agricultural knowledge, Science and Technology for Development (IAASTD) (2009). *Agriculture at a Crossroads*: *Global report 2009.* FAO, GEF, UNDP, UNEP, UNESCO, the World Bank and WHO.
http://www.unep.org/dewa/assessments/ecosystems/iaastd/tabid/105853/default.aspx

[2] Foresight. *The Future of Food and Farming* (2011) Final Project Report. The Government Office for Science, London.
https://www.gov.uk/government/uploads/system/uploads/attachment_data/file/288329/11-546-future-of-food-and-farming-report.pdf

In parallel with the promotion of the sustainable intensification concept, we observed that there has been a rise in interest in agro-ecology, as exemplified by a number of recent high level reports[3,4]. We therefore felt there would be merit in developing an understanding of the relationships between the sustainable intensification and agro-ecology concepts, the extent to which they are compatible, and whether or not agro-ecological systems and practices can form a valid path for achieving sustainable intensification.

While mindful of the political and social dimensions inherent in the concept of agroecology, we chose to focus on systems and practices as these can be used to support the management of individual farms. As a result, this particular report presents evidence from a desk-based review of agro-ecological systems and practices followed by an evaluation that compares agroecological and conventional systems in terms of energy and GHG emissions, biodiversity, soil and water, profitability and productivity. We are now of the opinion that agroecology can form an integral part of sustainable intensification, although there are a number of barriers hindering the wider adoption of this kind of approach, in particular those relating to knowledge exchange.

Clearly, further work is required to improve our understanding of the opportunities for agroecological systems and practices to contribute to sustainable intensification - and for these to be more widely adopted on farms. We now very much hope that others will use the work we have commissioned in order to inform further research on this topic. In particular, we believe that knowledge-based systems have a significant role to play alongside that of technology – with substantial benefits to be gained from working with nature as part of a more sustainable approach to agricultural production.

R. Jenkins

Ruth Jenkins

Chair of the Land Use Policy Group

---

[3] The Centre for Agroecology and Food Security (2013) *Mainstreaming Agroecology: Implications for Global Food and Farming Systems*. Coventry University/Garden Organic http://www.coventry.ac.uk/Global/05%20Research%20section%20assets/Research/CAFS/Publication,%20Journal%20Articles/MainstreamingAgroecology_WEB.pdf

[4] Olivier de Schutter (2011) *Agroecology and the Right to Food*. Report by the Special Rapporteur on the Right to Food. United Nations. http://www.srfood.org/images/stories/pdf/officialreports/20110308_a-hrc-16-49_agroecology_en.pdf

ACKNOWLEDGEMENTS

The Organic Research Centre and the Game and Wildlife Conservation Trust gratefully acknowledge:

- the extensive contributions made by external experts contacted by the team as part of the review process;
- the constructive input and feedback of members of the steering group (Cécile Smith and Maria de la Torre, Scottish Natural Heritage; Brian Pawson, Natural Resources Wales; James Petts, Natural England; and Steve Aston, Defra), both during steering group meetings and through extensive comments on the draft reports;
- the helpful and encouraging inputs from the external reviewers (Prof. Tim Benton, University of Leeds; Prof. Ian Crute, Non-Executive Director, AHDB; Prof. Charles Godfray, Oxford University; Stuart Knight, NIAB; Daniel McGonigle, Defra; and Prof. Christine Watson, SRUC), both during the workshop on January 14th, 2015, and subsequently;
- the active participation of attendees in the January 2015 workshop; many of whom also provided detailed contributions after the event;
- the authors and publishers who have granted permission for the use of Figures in this publication;
- and last but not least, all the contributors from our staff:
  - At ORC: Nicolas Lampkin; Bruce Pearce; Henry Creissen (at Teagasc from April 2015), Catherine Gerrard, Robbie Girling (at Reading University from March 2015); Susanne Padel; Jo Smith; Laurence Smith; Anja Vieweger; Martin Wolfe.
  - At GWCT: Alistair Leake; Sofi Lloyd; Phil Jarvis

Any errors or omissions are of course our responsibility as contractors.

SUMMARY

Background

'Sustainable intensification' is now often used to describe the future direction for agriculture and food production as a way to address the challenges of increasing global population, food security, climate change and resource conservation. While sustainable intensification is interpreted by some to relate to increasing production, with more efficient but potentially increased use of inputs and technology, there is also a need to consider environmental protection, including the conservation and renewal of natural capital and the output of ecosystem services. There is a growing consensus that sustainable intensification should not only avoid further environmental damage, but actively encourage environmental benefits. This includes addressing issues of consumption (including diets), waste, biodiversity conservation and resource use, while ensuring sufficient overall levels of production to meet human needs.

'Agroecology' is also now receiving increasing attention as an approach to agriculture that attempts to reconcile environmental, sustainability and production goals by emphasising the application of ecological concepts and principles to the design and management of agricultural systems. Agroecology can be seen as part of a broader approach to sustainable intensification focusing on ecological (or eco-functional) and knowledge intensification alongside technological intensification.

Main findings

This report explores how agroecological approaches can contribute to sustainable intensification in the UK and European contexts by:

- (i) exploring the concepts of 'sustainable intensification' and 'agroecology' (Chapter 2);
- (ii) reviewing the range of individual practices and systematic approaches that are typically defined as agroecological (Chapter 3);
- (iii) assessing the extent to which different agroecological approaches can contribute to sustainability outcomes (Chapter 4); and
- (iv) considering the policy drivers and constraints that may affect the adoption of agroecological approaches (Chapter 5).

Agroecological perspectives may be applied to the management of soils, crops and livestock, as well as to broader societal, environmental and food system issues. Agroecological practices, such as the use of rotations and polycultures, biological pest control, or legumes to biologically fix nitrogen, are not unique to particular groups of farmers. They can be used by all farmers, individually or in combination. However, synergies between individual practices can be important. Agroecology emphasises the idea of 'system redesign' rather than 'input substitution' for maximum benefit. In some cases, as in organic farming, the combination of practices may be codified (regulated) to enable marketing of products at premium prices to consumers. A range of more or less codified, systematic approaches, ranging from integrated pest and crop management through conservation agriculture and organic farming to agroforestry and permaculture, are described in the literature.

Three of the best documented approaches – integrated crop/farm management, organic farming and agroforestry – are assessed in detail, in comparison with intensive, conventional systems, with respect to their contribution to: (i) productivity; (ii) energy use and greenhouse gas emissions; (iii) biodiversity and related ecosystem services; (iv) soil and water conservation; and (v) profitability. This analysis concludes that agroecological approaches can:

- maintain or increase productivity, with the exception of organic farming where yields per ha may be substantially reduced due to restrictions on the use of agrochemical inputs – however organic system productivity with respect to other inputs including labour, and in terms of resource use (other than land) per unit of food produced, may be similar or better;
- contribute to reducing non-renewable energy consumption, both on a per unit of land and a per unit of product basis – although the benefits per unit of product are not as high in the organic case due to the lower yields;
- maintain or increase biodiversity and the output of related ecosystem services – with appropriately designed and managed agroforestry and organic systems offering potentially greater benefits than integrated systems;
- maintain natural capital in the form of soil and water resources as a result of reduced use, careful management (e.g. reduced or zero tillage) and reduced or restricted use of potentially polluting inputs;
- maintain or increase the profitability of farming systems through more efficient input use reducing costs, diversifying the range of outputs and, in the organic case, developing specialist markets with premium prices to help compensate for the lower yields.

The analysis further suggests that there will be both win-win situations, as in the case of agroforestry, as well as trade-offs between objectives, for example between productivity and biodiversity in the organic case. The latter might be compensated for by market mechanisms and/or policy interventions. To the extent that high outputs per unit land depend on inputs of non-renewable resources and degradation of natural capital, some compromises might be needed to deliver longer-term sustainability. This also illustrates the need for the maintenance of functional biodiversity components in productive agricultural landscapes to deliver the ecosystem services that can enable reduced use of unsustainable inputs and practices.

Overall, there is a clear case that agroecological approaches can make a substantial contribution to sustainable intensification, but this needs to be supported by an improved knowledge system (including training, education, advice and research with active farmer engagement), as well as by policy drivers, such as those adopted in the French agroecology action plan, to encourage change. There is also no one single approach that is likely to deliver all benefits simultaneously – a mosaic of approaches addressing specific needs is likely to deliver better overall results, as well as provide insurance against a single preferred strategy failing to deliver in practice.

On the basis of the analysis in this report, it is recommended that:

- Future work on sustainable intensification should place high priority on the sustainability component of the concept, including eco-functional and knowledge intensification, environmental protection and the delivery of ecosystem services;
- The potential of agroecological approaches to contribute to sustainable intensification (used in this sense described above) should be more widely recognised and developed. Agroecology is not just an option for, but an essential component of, sustainable intensification;
- Appropriate evaluation metrics should be developed to support business and policy decision-making, both at farm and regional/landscape level and taking account of different priorities (e.g. water use) in different areas;
- Policies to mitigate the negative impacts of many agricultural inputs, including fertilisers, pesticides, anti-microbials and anti-helminthics, should emphasise agroecological approaches in addition to technological or risk management solutions (as in the EU Sustainable Use of Pesticides Directive and the French agroecology action plan);
- Agri-environmental support, payments for ecosystem services (PES) and market-based policies (e.g. product certification) should be used to encourage the adoption of a broad range of agroecological approaches;

– Improved agroecological information and knowledge exchange systems, building on tacit farmer knowledge and active producer participation, should be developed and promoted. Achieving this will require better integration and co-ordination between individuals and organisations working on the subject, as well as the collaborative development of both on-line resources and traditional extension services;
– Educational provision, whether at vocational skills, further and higher education levels or more widely, should include a stronger focus on agroecological approaches – in the short term this issue can be addressed through the provision of targeted support (using the RDP vocational skills measures) but in the longer term a wide range of educational curricula need to be reviewed and updated;
– Research and innovation policy should include more focus on the development of agroecological approaches, not just their comparative evaluation. Support policies need to facilitate participatory delivery models and address the challenges involved in securing private sector funding for applied research that generates public knowledge not linked to saleable technologies and intellectual property.

Table of contents *Page*

1 INTRODUCTION

The Land Use Policy Group (LUPG) of the UK environmental, conservation and countryside agencies identified a need to develop a better understanding of the relationship between the concepts of sustainable intensification and agroecology, the extent to which they are compatible, and whether agroecological practices, systems and strategies are a valid and/or necessary path to sustainable intensification.

The Organic Research Centre (ORC), with the support of the Game and Wildlife Conservation Trust (GWCT), undertook this project to consider these questions for Scottish Natural Heritage (SNH), acting on behalf of the LUPG. The objectives of the study were to:

- describe what agroecological systems and strategies exist in both a UK and a European context;
- explore the relationship between sustainable intensification and agroecological approaches.

The study is intended to help to develop the work of the LUPG and others on sustainable intensification and follows research carried out for the LUPG by ADAS/Firbank (Elliott *et al.*, 2013) exploring the concept of sustainable intensification.

The current study is desk-based and appraises whether agroecological systems and techniques have relevance to sustainable intensification. It involves a systematic comparison of the relative performance of agroecological and conventional agricultural systems, based on a literature review and contact with experts.

1.1 Background

Against the backdrop of increasing world population, urbanisation and changing consumption patterns, the global demand for food is projected to increase. This anticipated increase in future demand comes with the imperative to safeguard the ecosystem services and natural capital underpinning agricultural production. This has led to the term "sustainable intensification" (SI) becoming widely used in the discourse on the future of food and farming, most recently in the Government's Agri-tech Strategy (HM Government, 2013), although the concept is subject to ongoing and intense debate (Garnett and Godfray, 2012; Huxham *et al.*, 2014).

The complexity of the issue is highlighted by the Government's Foresight study (Foresight, 2011), which recognised the need to reduce waste, reduce demand for resource-hungry products and protect biodiversity and ecosystem services, as well as to increase agricultural productivity. As stated in the final report:

"It follows that if (i) there is relatively little new land for agriculture. (ii) more food needs to be produced and (iii) achieving sustainability is critical, then sustainable intensification is a priority. Sustainable intensification means simultaneously raising yields, increasing the efficiency with which inputs are used and reducing the negative environmental effects of food production. It requires economic and social changes to recognise the multiple outputs required of land managers, farmers and other producers and a redirection of research to address a more complex set of goals than just increasing yield".

The BBSRC working group on the sustainable intensification of agriculture (BBSRC, 2014) has been grappling with some of these issues, concluding that there should be an equal and joint emphasis on 'sustainable' and 'intensification'; that just producing more with less is not enough; and that a holistic, interdisciplinary and systems-based approach is required. This requires better definitions and measurement of sustainability; more appropriate soil and land management; greater resilience to biotic and abiotic stresses and more effective and sustainable use of resources.

In an attempt to address some of the challenges identified in the SI debate, Defra has recently established the Sustainable Intensification Research Platform (SIP) (Defra, 2014a). The aims of the platform are to:

- develop, test and demonstrate systems-based, integrated approaches to UK land management that increase productivity and profitability while reducing environmental impacts, enhancing biodiversity and delivering wider ecosystem services;
- establish a platform to bring together researchers, policy, industry and other stakeholders to better coordinate sustainable intensification research.

The SIP intends to provide robust evidence to inform the farming industry's drive for sustainable intensification by researching integrated and collaborative approaches that enable farmers to increase food production and farm profitability while improving their environmental performance. It aims to host and link future agricultural research funded by Defra and other organisations and comprises of a physical network of agricultural study sites in England and Wales, and a community of collaborating agricultural, environmental and socio-economic researchers from over 30 organisations.

As a contribution to the SI debate, the LUPG commissioned and published a report that produced quantified evidence, using farm level case studies, of the environmental and production gains occurring in situations where farm management is thought to be consistent with a "sustainable intensification approach" (Elliott *et al.*, 2013). The report's authors made clear that the outcomes should be viewed as a pilot study, identifying some of the principal issues involved in designing and implementing a system for collecting and interpreting the kinds of farm-level data needed to quantify the strategic changes taking place within UK agriculture. They concluded that:

- The project had not revealed an overall negative relationship between food production and other ecosystem services. However, this may be because the set of variables was too limited and it may be appropriate to restrict comparisons to within farm types;
- It was possible to assess the sustainable intensification using individual farms' data already available and there was evidence that sustainable intensification has been practised by some farms in the UK in recent years;
- The main driver for farmers is profitability of their business but this is often delivered alongside improved environmental performance. However, a number of farms which increased food production also saw an adverse impact on environmental quality, indicating a trade-off between the two;
- Different farm types and systems have different potential for sustainable intensification. There may be limitations for the livestock industry and SI may not be appropriate in the uplands, where other ecosystem functions may have a greater social value than increases in food production;
- Actions to enhance and maintain biodiversity are largely a cost to the farm business, and sometimes require external financial support for their continued maintenance. The default approach appears to involve the use of the least productive land on the basis of least cost to the business, rather than decisions being informed by the best possible environmental outcome.

Overall the study found that sustainable intensification can, in some situations, be implemented in practice to deliver improvements in productivity while reducing environmental impacts. The concept, as demonstrated in the study and debated more widely, remains centred largely on the more efficient use of external inputs, advanced technologies and high levels of capital.

In parallel to the focus on sustainable intensification, there has been increasing interest in agroecological approaches, as exemplified in a number of high level reports, including from the Royal Society (2009), IAASTD (McIntyre *et al.,* 2009), OECD (2011), the EU Standing

Committee on Agricultural Research (SCAR, 2011), the UN (Schutter, 2010), UNCTAD (2013) and the RISE Foundation (Buckwell *et al.*, 2014). All these reports have recognised that agroecological as well as technological solutions such as genetic modification (GM) have relevance, with varying degrees of emphasis on the alternative options. Some (including Schutter (2010) and McIntyre *et al.* (2009)) have questioned the relevance of further technological intensification when much of the world's food is still produced by resource-poor, low-intensity farmers and argued that agroecological approaches have greater potential in such situations as they focus on using ecological understanding to make better use of existing resources and support producer autonomy and resilience.

Within the EU, there is an ongoing drive to reduce the impact of pesticides on the environment. This increasingly prioritises non-chemical (in part agroecological) approaches to agricultural pest and disease management. The EU Directive on the Sustainable Use of Pesticides (EC, 2009) makes it an obligation for Member States to (i) establish National Action Plans aimed at setting quantitative objectives, targets, measures, timetables and indicators to reduce risks and impacts of pesticide use on human health and the environment, and (ii) encourage the development and introduction of integrated pest management and alternative approaches or techniques in order to reduce dependency on the use of pesticides. In the UK, this has been implemented in the form of the Plant Protection Products (Sustainable Use) Regulations 2012 (SI 2012:1657) and the UK National Action Plan (NAP) for the Sustainable Use of Pesticides (Plant Protection Products) (Defra, 2013b).

Similar policy debates have been taking place in other European countries with some like France and Germany placing a strong emphasis on agroecological options. In the case of France, an agroecology action plan is being implemented (MAAF, 2014a,b), while in Germany the Bundesprogramm Oekologischer Landbau und andere Formen nachhaltiger Landwirtschaft (BOeLN); Federal Scheme for Ecological (organic) and Sustainable Agriculture) covers similar ground.

All of the issues, reports and policy drivers identified in this section are relevant to the debate on the potential contribution of agroecology to sustainable intensification which is the focus of this report, with many aspects considered in more depth in subsequent sections.

1.2 Approach

This report is divided into three main sections: the first focuses on the definitions of sustainable intensification and agroecology; the second on a literature review of agroecological concepts, practices, systems and strategies, and the third on an evaluation of their contribution to sustainable intensification. On the basis of this analysis, conclusions are drawn with respect to research, development and policy needs for enhancing the role of agroecological approaches in supporting sustainable intensification.

1.2.1 Literature review of agroecological systems and strategies

The literature review considered evidence from a range of bibliographic databases and specialist collections such as the international research database www.orgprints.org. Both peer-reviewed and research/evidence-focused grey literature were utilised, as well as communications with researchers in the UK and elsewhere. The review also draws upon the proceedings of recent conferences such as the FAO agroecology conference in Rome in September 2014[5] and the OECD/ASA organic farming conference in Los Angeles in November 2014[6]. We have also included literature from French and German agroecology

[5] http://www.fao.org/about/meetings/afns/en/
[6] https://www.agronomy.org/membership/communities/organic-management-systems

and organic farming research, much of which is not widely available in English, including the 600+ projects funded under the German Government's BOeLN programme (see above), which were evaluated and summarised in previous work (Vieweger *et al.*, 2014).

The literature review is divided into two main parts. The first considers system components and practices, ranging from soils through to plants, animals and humans. It covers relevant agroecological processes, ranging from microbial to crop, farm, habitat and global levels, focusing on technologies and practices that can be classified as 'agroecological' or could be considered to form part of agroecological strategies in UK and European contexts. This includes a range of geographical, climate and farm type situations, from horticultural and arable cropping to mixed and pasture-based livestock systems, and from Mediterranean to northern European zones. The second part of the review looks at how these practices are integrated into systematic approaches such as integrated crop management, organic farming, agroforestry and permaculture. The theoretical considerations are interspersed with short examples to help illustrate some of the ideas presented (Chapter 3).

We have endeavoured to ensure that the selection of literature and evidence is consistent with government standards for evidence, for example those of Natural England[7], with its emphasis on relevance, completeness, accuracy and timeliness. Where information was lacking or incomplete, we have noted the extent to which this constrains our analysis, and we have identified evidence gaps and any uncertainties with or limits to the evidence base. However, because of (i) the very wide range of topics covered, and (ii) the need to rely in part on previous work on this topic due to resource constraints, the literature review taken as a whole does not claim to meet the standards of a systematic review with clearly defined evaluation questions and search terms.

1.2.2 Evaluation of performance

The evaluation of performance focuses on the potential contribution of agroecological practices and systems to sustainable intensification with respect to five key areas: agricultural productivity (in particular with respect to land and labour); profitability; energy use and greenhouse gas (GHG) emissions; soil and water conservation; and biodiversity and related ecosystem services. The evaluation draws on the literature reviewed, as well as quantitative (financial, productivity and environmental) data on some well-defined and/or regulated agroecological approaches such as organic and integrated farming. It was not, however, within the scope of the study to undertake primary data collection, experimental or modelling work directly. The evaluation is thus based entirely on the analysis of secondary sources, focusing on priority indicators identified in relevant studies (Chapter 4).

1.2.3 Discussion, conclusions and recommendations

Based on the literature review and performance evaluation, the discussion addresses the following key questions:

i. the relationship between the agroecological and sustainable intensification concepts;
ii. whether agroecological systems and strategies can contribute to sustainable intensification in UK and European contexts;
iii. the extent of any such contribution, i.e. whether agroecological systems can contribute to sustainable intensification as a whole; or whether only some components of agroecological strategies can do so, or whether the two concepts are broadly incompatible;

[7] http://www.naturalengland.org.uk/ourwork/evidence/default.aspx

iv. the extent to which agricultural policy drivers could affect the relationship between sustainable intensification and agroecology;

v. any opportunities and barriers to the wider adoption of whole systems approaches and practices that may form part of agroecological strategies within the UK and Europe.

1.2.4 Expert review

The draft conclusions of this report were reviewed at an expert group meeting held on 14th January 2015 in London to discuss the findings and issues the study raises. During this review, the broad conclusions were well received. The comments made during the meeting, and the several written responses received after the meeting, have been integrated into the final version of the report. The reviewers highlighted the need to find:

- specific solutions relevant to specific situations, rather than one-size-fits-all generic prescriptions;
- better ways of communicating information on agroecological approaches to farmers (individually and in groups) – the information needs to be relevant to individual farm circumstances;
- better metrics that can be used to evaluate performance of different systems with respect to a range of sustainability outcomes;
- better policy frameworks to support the change process.

These and other issues raised at the expert group meeting are considered in greater detail in Chapter 5 of the report.

2 DEFINING SUSTAINABLE INTENSIFICATION AND AGROECOLOGY

2.1 Sustainable intensification

The term 'sustainable intensification', first promoted by Pretty (1997), has been used increasingly in the context of UK agricultural policy in recent years. However, it is subject to a wide range of interpretations, leading to potentially differing conclusions as to the relevance of agroecological approaches. The concept, as demonstrated by the LUPG's report on sustainable intensification (Elliott *et al.*, 2013) and debated more widely, can be highly focussed on the use of external inputs, advanced technologies and high levels of capital. This is, to an extent, reflected in the UK Government's Agri-tech Strategy (HM Government, 2013). Some of the current interpretations, however, contrast starkly with Pretty's original agroecological vision of an agriculture '*relying on the integrated use of a wide range of technologies to manage pests, nutrients, soil and water. Local knowledge and adaptive methods are stressed rather than comprehensive packages of externally-supplied technologies. Regenerative, low-input agriculture, founded on full farmer participation in all stages of development and extension, can be highly productive*.'

While some authors place most emphasis on intensification to increase production, the environmental impacts of such intensification are also of concern to many. They argue that the sustainable intensification concept is more complex than a focus on technology and inputs suggests, and that a simple 'producing more with less' definition is inadequate. For example, Elliott *et al.* (*op cit.*) suggests that sustainable intensification can be implemented in practice to "*deliver improvements in productivity while reducing environmental impacts"*, or at least causing no increase in adverse environmental impacts. The Royal Society's (2009) report on sustainable intensification emphasised the use of science to "*increase production while at the same time protecting societies, economies and the environment from negative side effects"*. The Foresight (2011) *Report on the Future of Food and Farming* defines sustainable intensification as "*simultaneously raising yields, increasing the efficiency with which inputs are being used and reducing the negative environmental effects of food production*". A similar position is taken by the BBSRC working group on sustainable agriculture (BBSRC, 2014).

Another challenge in defining sustainable intensification is the distinction between increasing production and improving productivity, where productivity implies more efficient use of resources, including land, labour, water and energy. While the focus is often on yields per hectare, the challenge may not just be how much food we can produce per unit of land. Other resources may be more limiting, with soil degradation, water availability, oil and phosphate reserves all potentially key issues. If both total production and productivity are to be enhanced, does this mean more efficiently (per unit of output), but possibly still with an increase in the total use of material and technological inputs, or could ecological or eco-functional intensification play more of a role to help reduce input use overall?

The EU Standing Committee on Agricultural Research (SCAR) raises the question of whether we should be aiming to increase production at all, presenting contrasting productivity and sufficiency narratives (SCAR, 2011), and highlighting issues of over-consumption and waste. Sufficiency was defined as yields that can provide a diet nutritionally sufficient in terms of nutritional energy (2,800 kcal/person/day), protein and fat supply (Erb *et al.* (2009) cited in SCAR (*op cit.*). Some argue we are already capable of producing sufficient food for an increased global population if we use all the crops produced to feed humans directly (e.g. Smil, 2000; Hanley, 2014). The sufficiency concept could be extended to include vitamins and minerals, or to refer to the number of people fed per hectare (Cassidy *et al.*, 2013) as an indicator of productivity (an issue which we explore in more detail in the productivity section of Chapter 4). It could also be extended to include other provisioning services, such as the production of fibre, fuel and timber, or even the production of biodiversity and related ecosystem services, but this may go further than most definitions of sustainable intensification envisage. However, there may be situations, for example in the uplands, where the management of biodiversity and production of ecosystem services (such as maintenance of carbon stocks, water quality, water storage and flood risk management, as well as the contribution of cultural landscapes and wildlife to tourism and recreation) outweighs the production of food in terms of meeting human needs, with all the economic challenges that this represents.

The complexities and competing priorities of the sustainability and sustainable intensification concepts are reviewed in more depth by Garnett and Godfrey (2012), the Foresight review (Foresight, 2011) and, more recently, by the RISE Foundation report on sustainable intensification (Buckwell *et al.*, 2014). These reports recognise the need to reduce waste, reduce demand for resource hungry products and protect biodiversity and ecosystem services, as well as to increase productivity. Building on these perspectives, Garnett *et al.* (2013) suggest a starting point for sustainable intensification (SI) might be: "*increase(d) food production from existing farmland in ways that place far less pressure on the environment and that do not undermine our capacity to continue producing food in the future*" but go on to say "*SI is a new, evolving concept, its meaning and objectives subject to debate and contest*".

Buckwell *et al.* (2014) question whether, given the already high intensity of much of European agriculture, there remains significant scope for further physical intensification and suggest instead that the key concept should be knowledge intensification rather than the intensification of technological or other inputs. Given the wide variation in agricultural contexts across Europe, decisions on intensification need to be made at a local level, taking into consideration a range of local conditions and likely outcomes including impacts on the environment.

Based on these various arguments, the ideal approach might involve routes to sustainable intensification where greater environmental benefits[8] and increased production/productivity can be achieved simultaneously. This concept is illustrated in the previous LUPG report (Elliott *et al.*, 2013), where Figure 2-1 below demonstrates a conceptual relationship between food production and other ecosystem services. The line represents the current limits to combined outputs of food and other ecosystem services, implying that over much of the range, these outputs can be considered to be competitive with each other. However, over parts of the range (near each axis) the combinations are complementary with potential synergies.

This model also illustrates a range of potential development paths:

a. increased yield with no additional impact on the environment (the basic SI concept);
b. an increase in both food and ecosystem services (the ideal outcome?);
c. ecosystem service improvements plus some yield increases and
d. ecosystem service improvements with no increases in yield.

Buckwell *et al.* (2014) take this concept further and illustrate the trade-offs between environmental goods and agricultural yields as well as the potential for increasing both simultaneously. Using biodiversity as the environmental good (Figure 2-2), they show both the range of biodiversity outcomes consistent with a given yield level (and vice versa), and the different routes (arrows) to sustainability (extensification, intensification, sustainable intensification), as well as a 'biodiversity-yield frontier'. This frontier represents the maximum possible combinations of yield and biodiversity, due to the variability of production

[8] including increased production of ecosystem services and maintenance of natural capital as emphasized in the 2011 UK National Ecosystem Assessment report '*Understanding nature's value to society*' and subsequent publications (http://uknea.unep-wcmc.org/).

systems and locations, and suggests that there are considerable opportunities to achieve sustainable intensification if current production systems, represented by the solid line, can move closer to the frontier. In this case, sustainable intensification is represented by the three middle arrows, representing a range of outcomes from more of one variable, keeping the other constant, to more of both simultaneously. Intensification (with reduced biodiversity outputs) as well as extensification (with reduced food outputs) are considered to fall outside the range of sustainable intensification options.

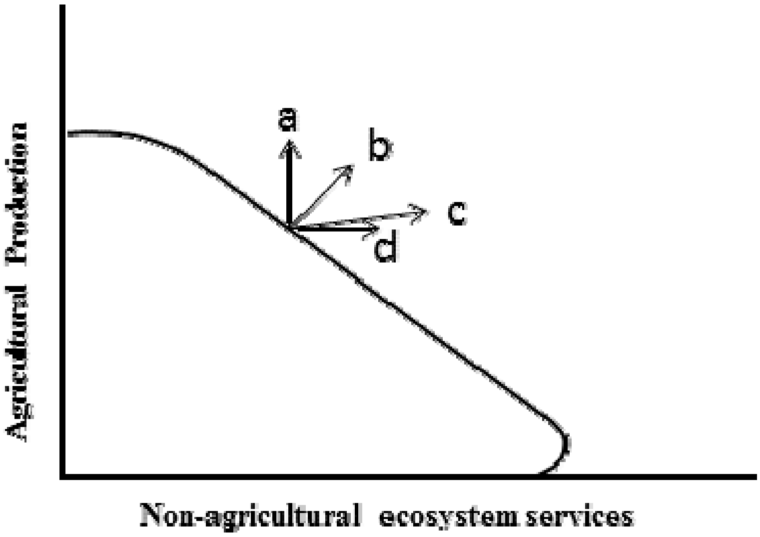


Figure 2-1: Concepts of sustainable intensification
Source: Elliott *et al.* (2013)

Figure 2-2: Sustainable intensification of biodiversity and yields
Source: Buckwell *et al.* (2014)

In our own study, three working definitions of sustainable intensification are deployed, reflecting the range of interpretations currently used in the debate:

1. producing more output, but with less resource use and environmental impact per unit of output, as in Elliott *et al.* (2013);

2. improving total factor productivity through improved resource efficiency and lowering environmental impacts per unit of output, while also delivering a range of other ecosystem services and enhancing the natural environment;
3. meeting human food, fibre, timber and fuel needs affordably, with sufficient yields, better utilisation (including diet), avoiding waste, improving (re)cycling and protection of renewable resources and reducing non-renewable resource use and waste.

In these definitions, it is recognised that: (i) output can be both physical yields and ecosystem services; and (ii) both efficiency and profitability are relevant and (iii) the substitutability (or renewability) of the resources used linked to the maintenance of natural capital needs to be considered. Capital-, technological- and/or eco-functional intensification are all potentially relevant contributors to the process.

2.2 Agroecology

Like the terms sustainability and sustainable intensification, 'agroecology' is used with a wide range of meanings in the current policy debate. In one sense, all agricultural systems might be considered agroecological in nature, given that they rely on biological processes and are conducted in an ecosystem context. Agroecology can also be understood in an academic discipline sense as the study of the ecology of agricultural systems, and used to describe ecological processes that operate in agricultural systems and the farmed environment.

However, the term agroecology has been popularised more as an approach emphasising ecological principles and practices in the design and management of agroecosystems, one that integrates the long-term protection of natural resources as an element of food, fuel and fibre production. The conceptualisation of this approach can be traced back to various authors in the 1970s, 1980s and 1990s, including Gliessmann (1998), Altieri (1995) and Mollison (1990), although it arguably has much earlier roots including the literature on organic farming from the 1930s onwards. More recently, agroecology has also been associated with radical social, economic and political perspectives, in particular linked to peasant agriculture movements such as La Via Campesina in Latin America (see, for example Pimbert (2009) and Wibbelmann *et al.* (2013)).

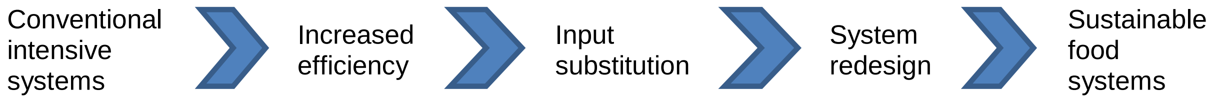
A key focus of such agro-ecosystem management approaches is increased reliance on knowledge and management (or design), complementing and reducing the use of technological inputs. For example, instead of using pesticides, the planting of refugia can be used to encourage natural predators to control insect pests. Key concepts in this context are those of diversity and complexity. Diversity, in the sense of using species and varietal mixtures as alternatives to monocultures[9], is now well understood to have positive effects on the control of diseases, pests and weeds, influencing both epidemics and the evolution of pest and pathogen strains resistant to control mechanisms. Mollison (1990) also highlights, as do other authors, the importance of complexity in system design. This includes particular emphasis on trophic dependencies (the reliance of organisms on their host for nutrient and other services) between system components, as well as the interaction between multiple components/ practices contributing to achieving particular goals such as weed or pest management, with each component/practice playing multiple roles within the system.

---

[9] Monocultures, or monocropping can be used in contrast to polycultures to refer to single species or cultivars grown on significant areas of land in a single year (as in this case), or continuously over multiple years, with or without occasional break crops. These terms are used in both senses in this report. The term polycultures is used to include the full range of combinations, both random (mixtures) and planned (e.g. intercropping as alternate rows or strips), of different species grown in combination.

There are a number of agricultural systems which are cited as being based upon agroecology and described in more detail at the end of Chapter 3. These include organic and biodynamic farming (with the added aspect of a regulated/certified market), agroforestry and permaculture. Other systems may adopt some agroecological principles and practices, while still using some conventional (chemical) inputs, such as integrated farming, conservation agriculture and low input sustainable agriculture (LISA). Hence, the adoption of agroecological approaches occurs at different points along a continuum, ranging from the uptake of some techniques and practices to the adoption of a whole system approach, as well as at a wide range of different scales. While much of the literature is focused on crop production, there are also many examples of the applicability of agroecology to livestock production.

The adoption of agroecological concepts can be thought of in terms of a development pathway from input-intensive industrial systems through to highly sustainable, ecological systems (Titonell, pers. comm., 2014). This builds on the efficiency, substitution and redesign framework posited by Hill (1985) and MacRae *et al.* (1989) amongst others.



Leaving open the questions of whether conventional intensive systems are the most unsustainable and whether the pathway to more sustainable systems is a linear one, as portrayed here, there is a broad consensus that achieving agricultural sustainability is a process or journey involving incremental steps or improvements. No agricultural systems can claim to be perfectly sustainable, given the multi-objective nature of the concept and the inevitable trade-offs when objectives conflict. However, from an agroecology perspective, a system redesign approach based on ecological principles is considered more likely to get closer to a sustainable end point. Sustainable intensification, if focused mainly on producing more with less, represents only the first step on the way. While some initiatives may encourage input substitution, for example replacing harmful pesticides with less harmful alternatives, this does not imply implementation of an agroecological approach.

In the European context, there has been only limited focus on the role which agroecological approaches could play in contributing to sustainable intensification, despite this being central to Pretty's (1997) original concept and later reflected in the Royal Society's (2009) report, to which Pretty contributed. That report identified four key principles for agricultural sustainability: persistence; resilience; autarchy (self-reliance); and benevolence (the ability to produce while sustaining ecosystem services and not depleting natural capital). The Royal Society report also argued that sustainable systems should exhibit most of the following attributes:

- utilise crop varieties and livestock breeds with high productivity per unit of externally derived input;
- avoid the unnecessary use of external inputs;
- harness agroecological processes such as nutrient cycling, biological nitrogen fixation, allelopathy[10], predation and parasitism;
- minimise the use of technologies or practices that have adverse impacts on the environment and human health;
- make productive use of human capital in the form of knowledge and capacity to adapt and innovate and social capital to resolve common landscape-scale problems;

[10] the chemical inhibition of one plant (or other organism) by another, due to the release into the environment of substances acting as germination or growth inhibitors.

- quantify and minimise the impacts of system management on externalities such as GHG emissions, clean water availability, carbon sequestration, conservation of biodiversity, and dispersal of pests, pathogens and weeds.

In France, the concept of 'Ecologically Intensive Agriculture' (EIA) is often used instead of sustainable intensification. This concept was highlighted during the Grenelle Environment Conference in 2007[11] and is now reflected in the French Government's commitment to agroecology, including an action plan launched in 2014[12]. Cassman (2008) defined ecological intensification as '*a process which increases yields and decreases agriculture's ecological imprint at the same time*'. This is similar to the UK definitions of sustainable intensification quoted above, assuming '*ecological imprint*' can be interpreted in the broad sense of environmental impact. The concept of ecological intensification as used in France and elsewhere has been more recently reviewed by Gaba *et al.* (2014), highlighting also its usage by UK authors including Garnett *et al.* (2013).

However, Pretty (*op cit.*) stressed that sustainable intensification policy should not prescribe '*specific, concretely defined technologies or practices, as this would restrict future farmer options. Farmers and communities should be allowed and encouraged to adapt to changing conditions; what needs to be sustainable are local processes of innovation and adaptation.*'

Crute (2015, pers. comm.) supports the view that sustainable intensification requires an inclusive, technology-neutral approach to a set of 'ends' rather than a preoccupation with 'means'. According to Crute, the concept should be outcome focused, with no one formula or vision and no technology ruled in or out. He argues that codification of practices, as in the case of organic farming regulations, should only be seen as a route to marketing the products. This technology-neutral position does not, however, address the environmental and other impacts of specific technologies or combinations of technologies, which we examine in Chapter 4.

For the purposes of this review, 'agroecology' is taken to mean the application of ecology to the management of agricultural systems at three levels of adoption:

1. an efficiency/substitution approach focusing on alternative practices and inputs with an emphasis on functional biodiversity, or eco-functional intensification, to reduce or replace external, synthetic, non-renewable inputs;
2. a whole system redesign approach focused on the farm ecosystem;
3. a focus on agriculture as a human activity system, including the issues of labour and knowledge/skills on farm as well as interactions between producers, supply chain actors and consumers.

For reasons of limiting the scope of this review, we have consciously chosen not to extend it to include agroecology as a social movement or to extend the system boundary to include entire food systems.

[11] http://www.developpement-durable.gouv.fr/La-genese-du-Grenelle-de-l.html
[12] http://agriculture.gouv.fr/IMG/pdf/plaqPA-anglais_vf_cle01abac.pdf see also MAAF (2014a,b)

3 AGROECOLOGICAL PRACTICES AND SYSTEMS – A LITERATURE REVIEW

The purpose of this literature review is to describe practices, technologies and systems considered to be agroecological or that could be considered to form part of agroecological strategies in the UK and European context. Consistent with the definitions of agroecology set out in the preceding chapter, we have divided this review into two main parts:

1. an overview and more detailed examination of individual agroecological management practices or system components, and;
2. a description of different system level approaches, which rely heavily on agroecological principles, such as integrated, organic and conservation agriculture as well as permaculture and agroforestry.

The review covers a range of geographical, climatic and farm type contexts, ranging from horticultural and arable cropping through to mixed and pasture-based livestock systems, and from Mediterranean to northern European zones. This contrasts with much of the existing agroecology literature which focuses on developing country situations, but we have deliberately chosen to emphasise the material which is relevant in the UK and EU land use context. Occasionally, some examples from other regions in the world, including the tropics, were used to illustrate particular concepts.

3.1 Agroecological management practices and system components

A wide range of agricultural practices and system components are identified in the literature as being agroecological in nature. The following list provides an illustrative overview, but is not exhaustive:

- reliance on soil biota, e.g. earthworms, for soil structure, formation of water stable aggregates, and soil water infiltration;
- biological nitrogen fixation using legumes and symbiotic N-fixing bacteria;
- the use of biologically active soil amendments (e.g. composts) to suppress soil-borne diseases;
- passive biological control of pests using field margin refugia or beetle banks to encourage presence of beneficial insects;
- temporal and spatial design of cropping systems to disrupt pest life cycles or attract pests away from sensitive crops (including push-pull systems);
- crop rotation to manage soil fertility and crop protection more generally;
- use of cultivar and species mixtures, including perennial and annual species and composite cross populations within species, to improve resource use efficiency and reduce pathogen spread between individuals with different genetic susceptibilities;
- utilisation of grassland by multiple livestock species, ensuring effective resource utilisation (different grazing behaviours) as well as health management (pathogen/parasite transfer and lifecycle patterns in pastoral ecosystems).

There are some common features within these practices:

- they have a strong biological rather than technological focus, with reliance on knowledge, skills and experience for their effective management;
- they emphasise diversity of system components and complex relations between components to deliver system resilience and stability;
- to the extent that they are used effectively, they permit reduced use of industrial/ technological/ synthetic agrochemical inputs.

Mollison (1990) describes the idea of complexity in agroecosystems as follows:

- each function (e.g. weed control) is delivered by multiple components/practices (e.g. variety selection, timing of sowing/planting, rotations etc.)

- each component/practice (e.g. green manures) has multiple functions (e.g. nutrient conservation, nitrogen fixation, soil protection etc.)

This builds on the ecological theory of niche differentiation - different species obtain resources from different parts of the environment, and the greater the number of trophic relationships (where one organism obtains resources from another), the more resilient a system is to shocks or disturbances that may impact seriously on one component. It is clear that any of these practices can be used by any farmer, but it is the use and integration of multiple practices and the possible synergies at a system level that characterises an agroecological approach to agriculture.

We explore these issues in more detail in the following sections.

3.2 Soil ecosystems and health

The soil is not just a physical growing medium providing anchorage for plants. It is a complex ecosystem with a wide range of (often) still poorly understood processes that provide nutrients and help control soil-borne pathogens and parasites. Ecosystems, including soil ecosystems, normally have the capacity for self-renewal. If this capacity is lost, specific indicators can be detected, such as soil erosion, a decline in fertility, changes in water holding capacity or species loss in the soil biota (Callicott, 1999). The concepts of soil resilience and health were fundamental to Balfour's "*Living Soil*" (Balfour, 1943). Leopold argued that the health of the land is tied directly to the integrity of the biotic community (Leopold *et al.*, 1949). He described its crucial influence on system stability and explored the idea of soil as an organism.

There are various definitions of soil health found in more recent literature, in relation to agricultural systems, primarily with a focus on its functional and productive aspect. For example, soil health is described as the "*capacity of the soil to support productivity and ecosystem services*" (Kibblewhite *et al.*, 2008); or the "*capacity of soil to function as a vital living system, within ecosystem and land-use boundaries, to sustain plant and animal productivity, maintain or enhance water and air quality, and promote plant and animal health*" (Doran and Zeiss, 2000).

In recent years, the degradation of soils in Europe and elsewhere in the world has led to a new focus on the protection of soils, highlighting the crucial role of soil as natural capital, delivering ecosystem services for the environment and the economy (DEFRA, 2009; Haygarth and Ritz, 2009; Dominati *et al.*, 2010; Dobbie *et al.*, 2011; European Commission, 2012; Wall *et al.*, 2012; Robinson *et al.*, 2013). However, the EU Soils Directive, which focused strongly on these issues and stimulated extensive debates, was not implemented due to opposition by some member states.

Concerns about soil degradation globally have also led to increasing the concept of soil security (Koch *et al.*, 2012; 2013), complementing similar food, water and energy security. Soil security refers to the maintenance and improvement of soil resources so that they can continue to provide food, fibre and freshwater, make major contributions to energy and climate sustainability, and help maintain biodiversity and the overall protection of ecosystem goods and services.

3.2.1 Nutrients and soil fertility

Effective nutrient management has always been an important component of maintaining productive and sustainable agricultural systems (Goulding *et al.*, 2008). Within this context, the availability of synthetic nutrients, in particular nitrogen (N), has contributed greatly to food supply and it is estimated that the food security of half of the world's population is dependent on fertiliser use (Global Partnership on Nutrient Management, 2010). Despite this, much of the fertiliser applied to crops is lost to the environment, in particular through nitrate leaching, volatilisation and denitrification (Lillywhite and Rahn, 2005). This leads to increased eutrophication and global warming through nitrous oxide (N_2O) emissions (the global warming potential of N_2O is 298 times greater than CO_2). The extent of the current losses highlights the need for further improvement in nutrient use efficiency within the scope of sustainable intensification.

3.2.1.1 Maintaining a closed system

Agroecological approaches emphasise the concept of a closed system to conserve resources, i.e. limiting the amount of nutrients imported to the farm or lost to the environment, through the adoption of more effective management systems and techniques. A key principle is to fertilise soils rather than crops, and to reduce nutrient losses by avoiding waste and unnecessary exports, for example potash in straw and forage (Lampkin, 1990; Watson *et al.*, 2002). This is achieved through practices such as using legumes for biological nitrogen fixation, the recycling and effective management of organic matter, for example by composting, green manures and timely incorporation. Crop genetic diversity forms an important part of this process, allowing for increased optimisation of nutrient availability and a more balanced nutrient flow, in addition to reduced losses per area of land (e.g. through the use of cover crops and intercropping (Vandermeer, 1992; Altieri, 2000)).

Agroecological systems such as organic farming and permaculture (see section 3.6) also place an emphasis on mixed production of crops and livestock, which allows for the utilisation of fertility-building crops not suitable for human consumption, co-operative use of farmyard manure between crop and livestock operations, and reduces the chance of stockpiled manure and slurry on livestock farms leading to leaching, emissions and other environmental problems (Scialabba and Müller-Lindenlauf, 2010). As with all practices that can be used by farmers, this approach is not unique to farms adopting agroecological practices, however there is a greater propensity to mixed farming within agroecological systems (Norton *et al.*, 2009; Ryschawy *et al.,* 2012).

The effective use of organic manures also supports biologically-active soil ecosystems. With organic manures, nutrients are applied to the soil together with organic matter, providing a source of energy (from the carbon compounds) for the soil ecosystem that is not available when mineral fertilisers are used. Soil organisms will use the nutrients in whatever form they are applied (mineral or organic) but they also need an energy source to respire and reproduce. With mineral fertilisers, soil organisms will break down existing soil organic matter, contributing to the decline in soil organic matter levels associated with intensive cropping systems (Boardman and Poesen, 2006). Despite this, a reliance on organic manures rather than soluble mineral fertilisers may lead to poor synchrony in the release of inorganic N and its uptake by the crop (Torstensson *et al.*, 2006).

At least in principle, closing cycles with respect to nitrogen and carbon is relatively easy, due to the atmospheric pools and biological fixation potential. This is because, although these nutrients may be exported or lost to the environment, they will sooner or later return to the atmosphere from where they can be sourced again using appropriate practices (see Figure 3-1 for the case of nitrogen). However, farmers still need to manage crop choice and utilisation as well as rotation design to achieve a balanced nutrient budget. This is potentially possible even on stockless organic farms (Smith *et al.,* 2014a), although integrated farmers and others may still rely on purchased, industrially-fixed nitrogen to meet some of their requirements.

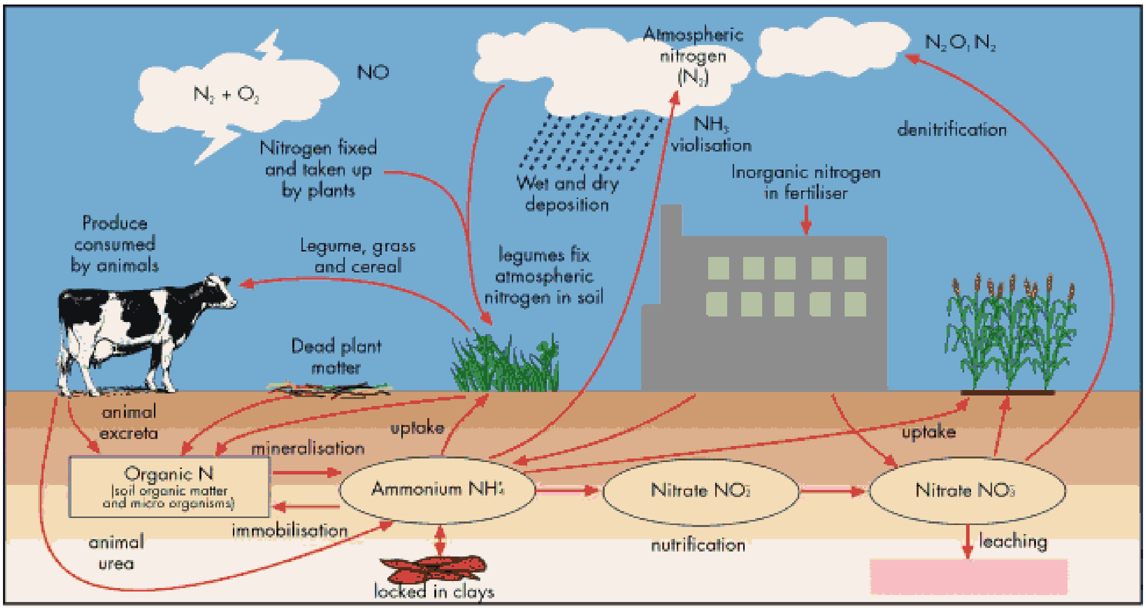
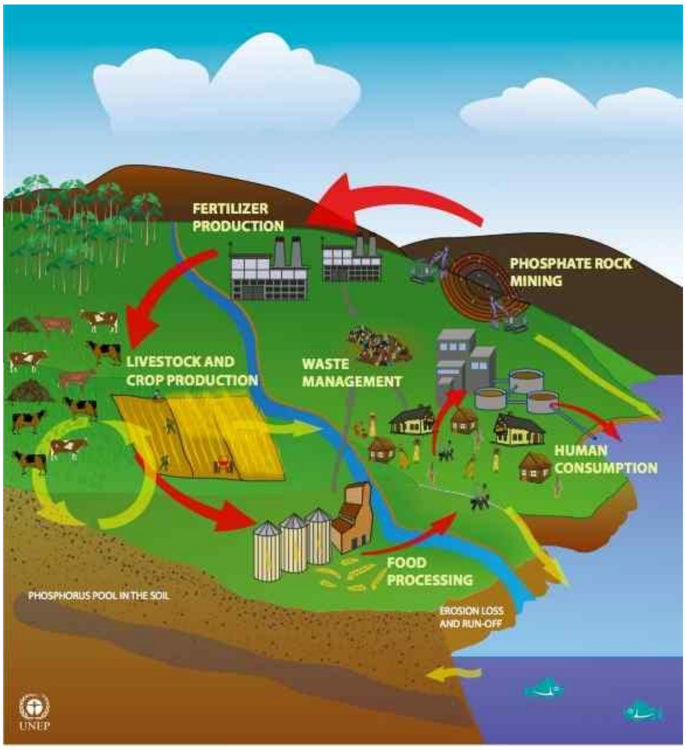


Figure 3-1: Nitrogen cycle showing significance of ammonium-N as a plant nutrient and first stage in breakdown of organic-N, a soil ecosystem process which is by-passed with the use of nitrate-N fertilisers
NB nutrification = nitrification and violisation = volatilisation
Source: Agricultural Bureau of South Australia (undated)
http://bettersoils.soilwater.com.au/module2/images/27.gif

The situation is more complex for other major and minor nutrients that are soil-bound and exported from the farm, either in products or through losses to the environment (e.g. via leaching or soil erosion). Particularly in the case of phosphorus, a linear process exists – from mining to on-farm application, harvest and sale, transport to urban areas, consumption and loss through sewage systems to marine environments (Figure 3-2).



There are increasing concerns about limitations to future global phosphorus supplies, similar to oil, and the potential of reduced phosphorus availability to limit responses to nitrogen fertilisers (Leifert *et al.*, 2009). Closing cycles, for example by returning harvested phosphorus and other nutrients from urban areas (see below), may need to be seen as something to be addressed on a regional rather than a farm level, and as a priority for all agricultural approaches.

Figure 3-2: Schematic illustration of linear approach to phosphorus utilisation in current agriculture. Source: UNEP (2011)

Although agroecological systems attempt to maintain a closed system, some have been shown to have phosphorus (P) and/or potassium (K) deficits following an assessment of supply and offtake. Within a study of nine organic farms, Berry *et al.* (2003) found that only those with large manure returns from stock utilising bought-in feed had a positive or neutral K budget. Gosling *et al.* (2005) also found that organic arable farms were mining reserves of P and K built up under conventional management and Korsaeth *et al.* (2008) and Torstensson *et al.* (2006) found P and K deficits within organic arable cropping and mixed dairy farming systems in Norway and Sweden. Some studies have found organic farms relying on manure inputs from conventional systems, in particular for the supply of P and K (Oelofse *et al.* 2010; Nowak *et al.* 2013: Oelofse *et al.*, 2013), even though such sources are restricted under organic regulations (Nowak *et al., op cit.*).

Despite this, stable balances can be achieved for P and K, without inputs imported from mined sources. This was demonstrated by Kaffka and Koepf (1989) in their study of N, P and K balance on a biodynamic dairy farm in southern Germany. The farm maintained relatively stable N, P and K levels over a 30-year period, with steadily increasing yields and soil organic matter levels, despite low levels of inputs. The reasons given for the effective maintenance of soil fertility in this situation included the:

- use of imported straw, supplying a source of P and K to the system;
- high rate of P and K recycling as a result of leaving straw and other crop residues in the field post-harvest;
- relatively low P and K content of the milk leaving the farm (i.e. the main product the farm produces);
- application of composts to the hay fields in the late summer when the plants are still growing so as to reduce N leaching;
- use of undersown clover and brassica catch crops following the cereal harvest to reduce leaching over winter;
- effective manure management (e.g. covering compost heaps and the use of soil dressing between layers of manure to reduce initial heating and subsequent N loss);
- use of a mixed farming system, incorporating cropping areas and livestock, thereby allowing for co-operative use of manures and utilisation of forages.

Expanding system boundaries to allow for closing of cycles beyond the farm could also help to improve P and K recycling (for example, through encouraging the use of treated sewage sludge or composted food waste on agricultural land). Significant progress has been made in the area of food waste recovery in recent years, thanks in part to the work of the Waste and Resources Action Programme (WRAP) (Quested *et al.*, 2013). The use of sewage sludge still faces barriers related to public perception and the UK's mixed sewage and wastewater systems give rise to potential heavy metal issues. The use of sewage sludge is strictly prohibited on organic land in Europe (Fonts *et al.*, 2012; Oelofse, 2013), although not in all agroecological systems. The organic prohibition represents a fundamental contradiction between the organic aspiration of closed systems and the need to maintain long-term soil fertility. Developments in the area of struvite (magnesium ammonium phosphate) recovery from waste water treatment plants could present a possible solution to this issue, also for non-organic farms, although this product is not currently on the list of permitted fertilisers within the EU Organic Regulation 889/2008 (EC, 2008).

Other options for closing cycles could include recycling of household organic wastes, either composted or as digestate following anaerobic digestion, or the spreading of green manures harvested from neighbouring land, either directly or in the form of digestate (Oelofse *et al.*, 2013; Stinner *et al.*, 2008).

Agroforestry systems can also promote more sustainable, closed systems with regard to the internal recycling of nutrients. Within agroforestry systems, nutrients are accessed and intercepted from lower soil horizons by tree roots and returned to the soil through leaf fall. Agroforestry systems thereby enhance soil nutrient pools and turnover and reduce reliance on external inputs. For example, leaf fall from 6-year-old poplars in an agroforestry system resulted in mean soil nitrate production rates in the adjacent crop-alley up to double that compared to soils located 8-15 m from the tree row, and nitrogen release from poplar leaf litter was equivalent to 7 kg N ha^{-1} yr^{-1} (Thevathasan and Gordon, 2004). Trees can also significantly influence nutrient additions to adjacent alley crops through intercepting rainfall (which contains dissolved, fixed nitrogen (see Figure 3-1)), via throughfall (rainwater falling through tree canopies) and stemflow (rainwater falling down branches and stems). Zhang (1999, in Thevathasan and Gordon, 2004), showed that these pathways contributed 11 and 15 kg N ha^{-1} yr^{-1} in hybrid poplar and silver maple systems respectively.

3.2.1.2 The role of legumes

As indicated above, the use of legumes for fertility-building in crop rotations is key to supplying nitrogen within agroecological farming systems, utilising the symbiosis between nitrogen-fixing bacteria and legume roots (Watson *et al.*, 2002). There are some exceptions to this, however, including the use of free-living *Azolla* to fix nitrogen in paddy rice systems. Legumes can be integrated in cropping systems as catch crops/green manures, polycultures (for instance, by mixing cereals with beans or peas) or as annual break crops and longer-term leys (mixtures of grasses, legumes and other species grown as herbage for livestock). Both grain and herbage legumes are relevant, although herbage legumes (such as clovers, lucerne, sainfoin, trefoil and others) are usually more effective than grain legumes at providing nitrogen to the subsequent crop (Dawson *et al.*, 2008). Non-grain legumes capture more carbon, leaving significant quantities of residual root biomass and helping rebuild soil organic matter levels in cropping sequences. A ley period within a crop rotation can also encourage healthier plants, by breaking soil-borne pest and disease cycles, thereby creating greater nitrogen use efficiency (Cook *et al.*, 1987). Within low-input conventional systems, the nitrogen uptake of wheat is greater after a legume crop than after another wheat crop or a fallow period (Soon *et al.*, 2001).

The fertility-building phase in a rotation can vary in duration depending on climate, soil and the suitability of the land for arable or horticultural crop production. The length of the ley period can vary from short term (12-18 months) to long term (around 5 years), but typically such leys are kept for between 18 months and 3 years. In Europe, organic farmers most frequently use grass-clover mixes for their leys. White clover *(Trifolium repens*) and red clover (*T. pratense*) are popular legume species while perennial ryegrass (*Lolium perenne*) and Italian ryegrass (*L. multiflorum*) are commonly chosen grass species. Most current leys are therefore relatively species-poor. The potential for nitrogen fixation by such leys is high, but the use of simple grass-clover mixtures can sometimes produce sub-optimal results owing to the cool, moist conditions required by both white and red clover species (Döring *et al.*, 2013). In addition, the use of red or white clover or other easily decomposable legumes as fertility building leys can lead to an asynchrony (mis-match) between the release of nutrients following incorporation of the green manure and the demands of the following crop (Crews and Peoples, 2005; Cook *et al.*, 2010; Dabney *et al.*, 2010; Campiglia *et al.*, 2011).

One way to improve the efficiency of the rotational system is to combine several legume species in a mixture, including a number of slower growing species, that results in a more complex residue structure with a better nutrient release profile. A recently completed Defra-funded LINK research project found that a mix of legume species can result in an increased (but variable) yield in addition to significantly increased ground cover, and reduced weed biomass, compared to monocultures. Benefits of mixing species with regard to productivity increased over time and stability of biomass production was greater in a mix containing 10 forage legumes compared to monocultures of a single legume species (Döring *et al.*, 2013). This project, and a related PhD project at Reading University (Brown, 2014), have also highlighted that legume species diversity can enhance the already proven benefits of legumes for pollinators (see below).

Use of white and red clover on conventional farms

White clover has become increasingly widely used on dairy and other farms in response to increasing nitrogen fertiliser costs. Red clover was once widely grown on UK farms but, in recent years, its use has declined significantly, except on organic farms. However, both white and red clover have been used to good effect on Adam Quinney's conventional sheep enterprise at Reins Farm, Worcestershire. The grassland derives all its nutrients from white and red clover and manure, thereby avoiding emissions associated with bought in fertiliser:

Nitrogen provided by clover nodules can help to reduce imports of synthetically fixed nitrogen and increase forage yields (Photos: ORC)

"*Red clover has transformed our system*," says Mr Quinney. "*All the lambs finish off it with no added concentrates. They grow at an average 0.34 kg/day, easily reaching 22 kg deadweight by the first draw at the beginning of October, when 80% are sent away. When we first introduced red clover, finished weights rose by 2 kg a lamb.*"

(Source: http://www.eblex.org.uk/wp/wp-content/uploads/2013/05/p_cp_down_to_earth300112.pdf)

In addition to the role of legumes in supplying N, leys and cover crops (with or without legumes) provide benefits for soil management through soil cover, nutrient capture and reduced soil disturbance. The use of cover crops has been shown to be effective at reducing leaching, through the immobilisation of N particularly on freely-drained, lighter soils, although the cost of establishment can be significant and interference with the following crop can lead to lower crop yields (Dabney *et al.*, 2001; Macdonald *et al.*, 2005). Non-leguminous crops can be used effectively for this purpose (e.g. mustard, *Sinapis alba* or forage rye). The N conserved by cover crops can then be released for uptake by the following crop through effective timing of incorporation (Watson *et al.*, 2002; Weinert *et al.*, 2002).

However legume-based green manures can also provide more N than the following cereal crop is capable of taking up, which can increase the rate of leaching (Dawson *et al.*, 2008). There are also risks that legume-based leys that are cut and mulched rather than removed or utilised by livestock, for example in stockless systems may also lead to increased nitrate leaching; but removal/use of the material, for example in biogas digesters, could substantially reduce the risks. There is a further risk when legume-based leys are ploughed in that a spike in nitrate leaching may occur, although studies of this phenomenon on organic farms still found average nitrate leaching levels to be similar or lower than on grassland farms using high levels of nitrogen (Phillips and Stopes, 1995; Stopes *et al.*, 2002). The nitrogen-use efficiency of some systems relying on manure and legumes can therefore be lower than those using imported, manufactured nitrogen fertiliser (Aronsson *et al.*, 2007).

Absolute N recovery will be reduced in systems with lower yields, but not necessarily proportionally, as over half of the nitrogen applied to growing crops in the UK is lost to the environment, leading to reduced drinking water quality, eutrophication and greenhouse gas emissions (Lillywhite and Rahn, 2005). However, in grassland livestock systems, grass-clover leys without applied nitrogen can outperform fertilised pure grass swards by a large margin in terms of N-use efficiency, with only a small reduction in absolute N-recovery, due to the high proportion of applied N that may be lost to the environment (Ryden and Garwood, 1984).

Sourcing fertility within the farm through the use of under-sown green manures:

Iain Tolhurst of Tolhurst Organic Produce in Oxfordshire has been making good use of clover and other green manures undersown within the growing crop to supply N, reduce losses from leaching, improve soil stability, promote functional biodiversity and control weeds. "*It is important to remember that greater losses of N can result from leaching than from crop removal*" says Mr Tolhurst. "*Through keeping the soil covered we are avoiding unnecessary losses while supplying N to the system, improving diversity and controlling weeds*". This organic system is managed without any livestock or inputs of animal manures.

(Photos: I Tolhurst)

Keeping the soil covered through the use of cover crops will help to reduce losses (Shepherd and Lord, 1996). Torstensson *et al.* (2006) found that the most efficient system was to apply mineral fertiliser combined with the use of a ryegrass cover crop. The use of precision application techniques within systems using mineral fertiliser can also help to improve nitrogen use efficiency by reducing inputs and losses (Cassman *et al.*, 2002; Diacono *et al.*, 2013). Spiertz (2010), however, highlights that such approaches tend to focus on improving N application methods, whereas considerable benefits can be obtained through improved system design (e.g. crop rotation planning to ensure effective sequencing of N supply and demand). Spiertz highlights the value of farmer benchmarking to improve efficiencies in this area, as this approach can help to highlight the economic benefits and environmental harm associated with specific practices. In addition, the alleviation of factors that reduce crop yields (e.g. pests and diseases, drought, flooding) can help to increase efficiencies across a range of systems by ensuring that optimum rates of N uptake are maintained at a farm system level.

Grain and forage legumes are not the only source of nitrogen-fixing potential in farming systems. There have been many studies assessing the value of green mulch from leguminous trees to enhance soil fertility for adjacent crops in tropical agroforestry systems (Yobterik *et al.*, 1994). However, relatively few of the 650 woody species that are able to fix atmospheric nitrogen occur in temperate regions. Of these, black locust (*Robinia*), mesquites (*Prosopis*), alder (*Alnus*) and oleaster (*Eleagnus*) have been investigated for their nitrogen-fixing potential (Jose *et al.*, 2004). Significant transfer of fixed nitrogen to crops has been observed in a study which showed that 32 to 58% of the total nitrogen in alley-cropped maize came from nitrogen fixed by the adjacent red alder (*Alnus rubra*) (Jose *et al.*, 2004). Red alder trees in the silvopastoral trial system at Henfaes near Bangor were studied to assess the potential for improving and maintaining soil fertility (Teklehaimanot and Mmolotsi, 2007). The rate of N fixation was estimated at 31 kg ha^{-1} yr^{-1} in the silvopasture treatment with tree densities of 400 stems ha^{-1} and the total amount of N that could potentially be added to the soil as a result of decomposition of dead leaves, roots and nodules was estimated at almost 41 kg ha^{-1} yr^{-1} (Teklehaimanot and Mmolotsi, 2007).

3.2.1.3 The role of soil microbial communities

As soil microorganisms perform many soil biological processes, the presence of an abundant and diverse soil microbial community is essential to sustain productivity of an agro-ecosystem (Altieri and Nicholls, 2003; Brusaard *et al.*, 2007; Janvier *et al.*, 2007; Wall *et al.*, 2012). Brusaard *et al.* and Wall *et al.* (*op cit.*) provide comprehensive overviews of the role of soil organisms in providing ecosystem services, and how this is affected by agricultural practices and agroecosystem management. Gardi and Jeffery (2009) also make reference to the role of the soil ecosystems and soil biodiversity in supporting pollinators.

To varying extents, plants actively regulate the microbe community in their rhizosphere through molecular communication and root exudates to suit their needs (Altieri and Nicholls, 2003; Berendsen *et al.*, 2012; Chaparro *et al.*, 2012). This is important both for symbiotic nitrogen fixation and for nutrient release from the breakdown of soil organic matter and the chemical weathering of soil minerals thanks to acids released by microbes. Several studies have recorded higher microbial diversity, increased enzyme activity and greater stability in agroforestry alley cropping systems, attributable to differences in litter quality and quantity, and root exudates (Seiter *et al.*, 1999; Lee and Jose, 2003; Mungai *et al.*, 2005; Udawatta *et al.*, 2008; Lacombe *et al.*, 2009).

The complex microorganism-communities of the plant, above and particularly below ground are crucial for plant health, and also referred to as the second genome of the plant (Berendsen *et al.*, 2012). Berendsen *et al.* explored the mechanisms that govern the ability of plants to 'recruit' specific microorganisms to fend off pathogens or pest attacks, highlighting the crucial importance of maintaining a high diversity of soil microorganisms in agricultural systems. Crop rotations, compost and organic matter additions (by adding energy and nutrients to the soil ecosystem) all influence the suppressive effect on soilborne diseases and pathogens by regulating and improving soil microbial communities (Larkin *et al.*, 2012).

As part of the soil microbiome, arbuscular mycorrhizal (AM) fungi penetrate the plant roots and form a bridge to the surrounding soil, enhancing plant nutrient uptake (in particular phosphorus), growth and health (Rillig *et al.*, 2002; Hijri *et al.*, 2006; Schädler *et al.*, 2010; Ortas, 2012). However, AM fungal diversity tends to be low in conventionally managed agricultural soils, which has been attributed to negative effects of fertilisation, fungicides, soil cultivations and low host diversity. It has been shown that low-input, conservation and organic systems may enhance AM fungal richness compared to conventionally managed systems (Mäder *et al.*, 2002; Chifflot *et al.*, 2009).

Observations from the long-term comparisons of conventional, integrated, organic and biodynamic systems in Switzerland, known as DOK, indicated an increase in a range of soil-borne organisms, including mycorrhizae, under organic compared with non-organic conditions (Mäder *et al.*, *op cit.*; Oehl *et al.*, 2004; Esperschütz *et al.*, 2007). However, the development of such approaches can take a considerable amount of time. Derpsch (2007) states that it can take over 20 years for the full benefits from the adoption of techniques promoted within conservation agriculture to be realised, with respect to internal nutrient cycling. The short term nature of many studies comparing the adoption of individual techniques used within conservation agriculture (such as reduced tillage and cover cropping) will therefore only present a snapshot of a system in transition from one type of management to another (Berner *et al.*, 2008). Annual crops are less well suited to the development of mycorrhizal associations (Dawson *et al.*, 2008), but perennial crops including grassland and woodland exhibit better potential, so that fertility-building leys in rotations, and agroforestry can help support AM colonisation.

New and recent research programmes are taking a fresh look at some of these issues, including the BBSRC/NERC '*Soil and rhizosphere interactions for sustainable agro-ecosystems'* (SARISA) programme, the Defra-led '*Soil security programme*' (SSP) and the EU-funded SOILSERVICE project '*Conflicting demands of land use, soil biodiversity and the sustainable delivery of ecosystems goods and services in Europe*'.

3.2.2 Soil conservation and reduced tillage

According to Hamza and Anderson (2005), the intensive use of heavy machinery, less diverse and short crop rotations as well as intensive grazing management are the main reasons for soil compaction being a major problem in modern agriculture. Apart from low soil organic matter content, soil compaction is caused primarily by high machinery traffic, particularly tillage, and intense animal treading in wet soil conditions (Hobbs *et al.*, 2008; Hamza and Anderson, 2005; Drewry, 2006). Alternate configurations of machinery tyres and tracks vary in their ability to generate tractive forces, and therefore in the way they cause soil disturbance, compaction and rut formation. This can potentially lead to reduced water infiltration rates and increased soil erosion and runoff (Raper, 2005). Hamza and Anderson (*op cit.*) further describe that soil compaction generally increases soil strength while decreasing soil fertility, by reducing storage and availability of water and nutrients; ultimately leading to higher fertiliser demand and increased production cost. Table 3-1 provides an overview of suggested practical methods and soil management strategies to avoid, delay or prevent soil compaction (Hamza and Anderson, 2005; Raper, 2005).

Table 3-1: Strategies to avoid and manage soil compaction

Machinery use	***Crop/grazing management***
Reducing pressure on soil (e.g. decreasing axle load, increasing contact area of wheels and soil)	Increasing soil organic matter through crop rotation (incl. plants with deep, strong taproots)
Avoiding working soil in high soil moisture conditions	Increasing soil organic matter through retention of crop and pasture residues
Reducing the number of passes by machinery and confining traffic to certain areas of the field (controlled traffic)	Maintaining appropriate base saturation ratio and complete nutrition to meet crop requirements to help the soil/crop system to resist harmful external stresses
Using conservation tillage systems which minimise vehicle traffic	Avoiding grazing in high soil moisture conditions
Sub-soiling to eliminate compacted soil profiles in crop growth zones	Reducing the intensity and frequency of grazing

Reducing tillage depth and frequency can help reduce compaction and N losses from farming systems, in particular losses from mineralisation and leaching, (Köpke, 1995). However, the adoption of the technique can have positive and negative effects, for example residues left on the soil surface can result in an immobilisation of N, resulting in a lower N availability for the growing crop (Dawson *et al.*, 2008) with resulting lower yields (Dou *et al.*, 1994).

Profile of a soil showing compacted layers on the left, and a spade sample of well-structured soil with high organic matter content right
(Photos: Left - Garden Organic, Right - P Sumption/ORC)

Reduced tillage is also a key component of conservation agriculture (see Section 3.6.3), which aims to promote long term soil fertility and slow release of nutrients for crop offtake (Kassam and Friedrich, 2009). Surface mulches and cover crops are also used in conservation agriculture, with the aim of emulating forest floor conditions. Cover crops act as 'nutrient pumps' within these systems to enhance and conserve pools of nutrients from which plant roots feed (Kassam and Friedrich, 2009). Nutrients are therefore concentrated in the biomass, and the first 10 cm of the soil, which can encourage uptake of a wide range of nutrients and trace elements (Habte, 2006). The lack of surface disturbance within conservation agriculture also allows for the development of mycorrhizal associations, which is normally curtailed through soil tillage and the intensive application of agrochemicals. These associations can encourage nitrogen and phosphorus uptake in situations of low supply (e.g. low input systems and degraded soils) and subsequent biological nitrogen fixation (fixation can be limited by the availability of phosphorus in situations of limited supply (Lynch, 2007)).

While the advantages of no-till or reduced tillage approaches, such as reduced energy use, CO_2 emissions and erosion, or increased soil fertility and soil biota activity/diversity, have been highlighted frequently (Berner *et al.*, 2008; Gadermaier *et al.*, 2011; Karlen *et al.*, 2013; Kuntz *et al.*, 2013), the difficulties with its adoption in low-input agricultural systems are mainly attributed to problematic weed control, and sometimes a decrease in yields (Dou *et al.*, 1994; Berner *et al.*, 2008; Krauss *et al.*, 2010; Drinkwater *et al.,* 2000)). Conversely, increases in yields with a reduced tillage approach were also found, for example by Mäder and colleagues, specifically for maize, winter wheat and grass-clover mixes (Mäder *et al.*, 2012) while Berner *et al.* (2008) found yields under organic conditions that were 97% of those obtained under conventional tillage. In addition soil organic carbon and microbial biomass were enhanced.

A key problem associated with reduced tillage in northern temperate climates is the damage caused to germinating cereal seedlings through seed hollowing by slugs. The presence of trash at the surface and the use of disc drills in minimum tillage systems can pre-dispose crops to attack. Increasing the drilling depth from 20 mm to 40 mm reduced this problem from 26% to 9% by excluding slug-seed access and by reducing the germination time after placing seed in moister conditions (Glen *et al.,* 1990). Residue management was a key topic which the Soil Management Initiative (SMI, 2005) found necessary to deal with in considerable detail during the transition from a plough-based system to adoption of a minimum tillage system (see also Section 3.6.2).

When comparing reduced tillage with conventional tillage in wheat and spelt crops over 3 years, Berner *et al. (*2008) were able to show an increase in soil organic matter by 7.4% in the 0-10 cm soil layer. They also found an up to 70% higher abundance of endogeic, horizontally burrowing adult earthworms under reduced tillage, compared to conventional tillage. Assessments of earthworm activity in the same long-term system comparison trial in Frick (Switzerland) nine years after its initial implementation, confirmed that total earthworm density under reduced tillage was significantly increased (Kuntz *et al.*, 2013). This effect was mainly attributed to a higher number of juvenile worms, the number of cocoons being five times higher under reduced tillage compared to conventional tillage.

Weed abundance, however, is usually more than doubled under reduced or no-till systems, and tends to remain at this level over time (Sans *et al.*, 2011; Armengot *et al.*, 2015). The work of Armengot and colleagues has shown further that abundance of perennial weeds increases over time, changing the species composition between tillage systems, which can be a challenge where no herbicides are used, or in some situations may lead to increases in herbicide use. In some respects these results are not surprising – using a fixed tillage approach continuously is likely to select a weed population that is adapted to that particular approach, and it may be necessary to alternate tillage systems to avoid certain weed problems building up.

Soil management is a key feature of agroforestry systems, and in both tropical and temperate climates, agroforestry systems are designed and implemented to counter soil erosion and degradation, and improve soil quality and health (Young, 1997). The replacement of natural forest and scrublands by croplands and grasslands devoid of trees on susceptible soils has resulted in increased run-off and accelerated erosion in many agricultural areas. When trees are reintroduced to the landscape as part of agroforestry systems many of these problems can be mitigated. Surface erosion is often reduced when tree canopies intercept precipitation and help reduce local rainfall intensity. The Pontbren project (in mid Wales) has demonstrated that tree planting on heavily grazed upland grassland helped to improve infiltration rates and reduce the 'flashiness' of adjacent streams (Woodland Trust, 2013). Tree roots can increase the structural stability of the soil and enhance water infiltration and improve water storage by increasing the number of soil pores (Hoogmoed and Klaij, 1994). Macropores (larger spaces between soil particles) rapidly channel surplus surface water flow and allow air and moisture to move into the soil. In this way the risk of soil erosion is reduced. Tree roots and trunks also act as physical barriers to reduce surface flow of water and sediment (Udawatta *et al.*, 2006; Udawatta *et al.*, 2008a). Tree roots prevent erosion and act as permeable barriers to reduce sediment and debris loading into rivers following floods. Planting *Populus x euramericana* on erodible slopes in New Zealand reduced pasture production losses due to landslides during a cyclonic storm by 13.8% (Hawley and Dymond, 1988, in Benavides *et al.*, 2009), while mature willow and poplar trees can reduce mass movement by 10-20% (Hicks, 1995, in Benavides *et al.*, *op cit.*).

3.3 Plants and cropping systems

Plants, crops and whole systems are subjected, continuously, to a wide range of challenges from the biotic and abiotic environment. The form and scale of such dynamic challenges are changing and increasing under climate change. Agroecological principles suggest that significant increases in diversity and complexity at all cropping levels, including within the crop, are needed to ensure the range of characteristics necessary to deal with environmental unpredictability (e.g. Østergård *et al. (*2009)).

3.3.1 Plant genetic diversity and breeding

Agroecosystem diversity is dependent on the range of crop and livestock species available, together with the varieties of those species. Over the last hundred years or so, the principal goal of plant (and livestock) breeding has usually been an increase in yield, often with an assumption that any deficiencies in performance due to pest, disease and other challenges, could be addressed by agrochemicals and machinery, albeit with a greater emphasis on resistance to pests and diseases more recently. In the agroecological approach, deficiencies in performance of individual varieties are primarily dealt with through system design, for example by improved rotations and greater overall diversity. This implies the need for specific breeding objectives to complement and address the limitations of agroecological systems and to increase their potential to deliver both high productivity and sustainability. Most agroecology advocates are critical of GM technologies, although some argue that there might be opportunities to combine agroecological and GM approaches. To date, there has been little focus on what might be termed agroecological breeding.

Some of the genetic characteristics required for cereals in agroecological systems were reviewed by Wolfe *et al.* (2008). These include resistance to seed-borne diseases, pests and weeds, and improved nutrient use efficiency from soil sources, including better mycorrhizal and soil bacterial relationships. They also identified that the range and importance of diseases, pests and weeds was generally different in agroecological systems compared with intensive conventional systems. Lammerts van Bueren and Myers (2012) expand this theme for all major crops.

Cereals in particular have seen a large increase in monoculture cultivation, with a consequent reduction in plant diversity. Negative consequences include a shift in the evolution of the yellow rust pathogen towards greater virulence and the ability to thrive over a wider range of climatic conditions (Milus *et al.*, 2009). One simple approach to increase diversity has been through the development and use of mixtures of up to 4 or 5 varieties (Wolfe, 1985), which saw its most large-scale expression in the German Democratic Republic in the 1980s, with the successful deployment of spring barley mixtures, designed specifically for control of barley powdery mildew, over some 3.5 million hectares (Finckh and Wolfe, 2006). The limitation of this approach, however, is that the level of diversity in the crops might still be too narrow to deal with the increasing range of environmental challenges. There are also issues associated with the marketability of mixtures for specific processing purposes as well as for use as seed crops. Grain consistency and quality issues have been raised by maltsters and millers, but some studies have shown that this need not be such an issue (Newton *et al.*, 2008). However, around half of the barley produced in the UK is used for animal feed for which grain consistency is not a major concern (Brown, 1995; Newton *et al.*, 2011).

Growing varietal mixtures of winter wheat to achieve stable yield and quality

Paul and Mark Ward who farm Priory Farm, a conventional low input farm in Suffolk, routinely grow a varietal mixture of winter wheat, which includes equal proportions of the modern varieties Deben, Alchemy, and Glasgow. Components of the mixture were originally selected on height differences to reduce competition for above ground space and light. For the last eight years, this mixture has achieved yields that are equal to or greater than those of the pure stands of modern varieties which are also grown on the farm. Most of the grain produced from the mixture goes for animal feed. In 2012, however, a year in which there were national issues with wheat quality largely due to a very wet summer, the mixture maintained its quality and went for biscuit making. This is a good example of farmers using within-crop genetic diversity to buffer against environmental stresses and achieve stable crop performance. (Photo: ORC)

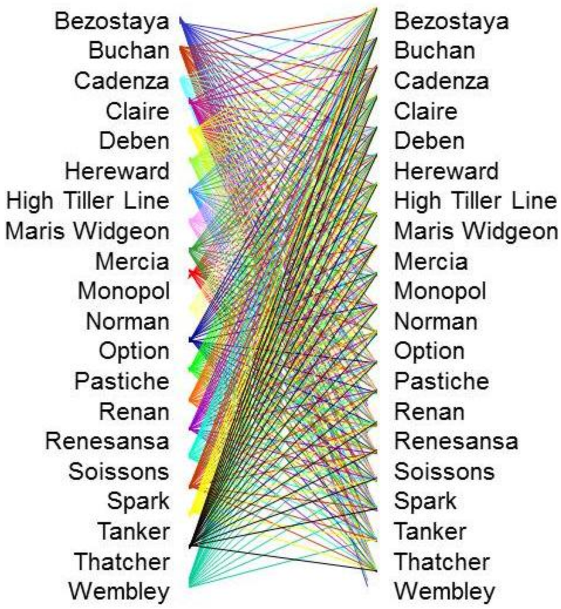
To address the environmental adaptability limitation of mixtures, an 'evolutionary breeding' approach (Suneson, 1956) has been utilised to produce composite cross populations of wheat and other cereals which incorporate high levels of genetic diversity. Such populations, in mimicking natural ecosystems, can provide increased resilience against complex physical and biotic challenges (Döring *et al.*, 2011), beyond simple varietal disease resistance and cold tolerance traits. A further advantage of the genetic variability of composite cross populations, is the potential for selection to thrive in different types of production systems, for example in mixed cropping or agroforestry. This will be dependent on an effective range of parental genotypes, but could provide a means of making better use of more complex and diversified cropping systems under the agroecological umbrella (Döring *et al.*, *op cit*.)

Within the context of a greater diversity of germplasm, another relevant issue is the potential of decentralised versus centralised breeding (e.g. Reguieg *et al.*, 2013; Horneburg and Becker, 2008). This refers to the degree to which a breeding programme should focus directly on the target area for the crop in question (principally the local physical environment) and the target agroecosystem (principally the local biotic environment), rather than on results obtained from a central breeding facility that produces a single product that is intended to perform across a range of different environments. Similar concepts lie behind the increasing interest in participatory breeding focusing primarily on open-pollinated species[13].

However, a significant disadvantage is the prohibition on marketing seed from these breeding programmes under current seed regulations as they do not meet variety uniformity and other requirements for registration. As a result of Defra and EU-funded work over the last 15 years led by the Organic Research Centre in collaboration with NIAB, John Innes Centre and others, winter wheat populations have been developed that have commercial potential (Girling *et al.*, 2014). Following negotiations with Defra and the European Commission, an EU marketing experiment is underway to investigate the marketing potential of populations, which if successful may lead to changes in EU seed regulations.

Consistent performance of winter wheat populations in Northumberland

Peter Hogg of Causey Park farm in Northumberland has been growing composite cross populations of winter wheat for the last five years. He was involved in on-farm trials conducted by the Organic Research Centre as part of the Defra-funded 'Wheat Breeding Link' project. Since then, he has continued to grow the Y (high yield parents), Q (high quality parents), and the YQ (all Y and Q parents) composite cross populations because they proved to be consistent performers. Overall the populations achieved yields of around 7 t ha^{-1} and in certain years produced great milling wheat while requiring lower nitrogen inputs compared to other milling wheat varieties grown on the farm.



The 20 parents and over 190 crosses that make the YQ population

[13] http://www.open-pollinated-seeds.org.uk/open-pollinated-seeds/Participatory_Plant_Breeding.html

3.3.2 *Plant spatial arrangements – rotations and polycultures*

For hundreds of years, crop rotation has formed the basis of UK agriculture, going back to the Roman 3-field systems, then the Norfolk four-course rotations and more recent alternate husbandry rotations involving periods of arable cropping followed by fertility-building leys (Lampkin, 1990). Crop rotations are usually more diverse in agroecological management systems compared to conventional systems; and often involve cover crops, green manures or intercropping (see below). In recent decades, the increased use of pesticides and fertilisers has meant that rotational constraints, for example fallows for fertility building and breaks between crops to control pests and diseases, could be overcome, allowing continual cropping of single species, sometimes with break crops to address specific requirements. However, increasing problems of weeds, pests and diseases resistant to chemical controls have led to renewed interest in rotational/ cultural controls, particularly for blackgrass (HGCA, 2014)[14].

Rotations provide a range of useful benefits including rebuilding soil fertility, in particular carbon and nitrogen fixation (Havlin *et al.*, 1990; Karlen *et al.*, 2013), the management of soil-borne diseases, pests and weeds (primarily by breaking pest and pathogen lifecycles), nutrient cycling and the option to integrate livestock production into farming systems.

Rotation design is itself a large and complex subject involving trade-offs between the desired agroecological impacts on the one hand, and the marketing and management objectives of the holding on the other. Verhulst *et al.* (2011) and Briggs (2008) highlight the importance of understanding the effects of rotation and residue management practices on plant growth and health, in order to choose suitable varieties, timing and management practices, as well as levels of external inputs (e.g. fertiliser).

Altering arable rotations to increase feed value and control weeds

John Pawsey of Shimpling Park Farm in Suffolk began the process of converting to a fully organic system in 1999. Changes had to be made to his conventional rotation of wheat, followed by rape, followed by sugar beet, to include more nitrogen fixing legumes and spring cereals. His rotation now consists of winter wheat, spring cereal, winter beans, spring cereal undersown with a red clover ley for one year. The rotation is currently being altered to extend the ley period for an extra 1 or 2 years, which will provide greater control of blackgrass in addition to producing more feed for the recently acquired sheep. The diversity of the ley will also be increased to include a greater number of legume species, which will improve feed value and provide additional benefits to pollinators as a result of an increased flowering period.


(Photos: J Pawsey)

[14] http://www.hgca.com/media/433525/is30-black-grass-solutions-to-the-problem.pdf

In nature, rotations occur only rarely, and on very different time scales. In this sense, rotations can be considered as artificial, mimicking the early stages of ecological succession, although they do result in known and positive agroecological impacts by providing diversity over time and, to the extent that this creates a mosaic of land uses at landscape level, some diversity in space as well.

Many of the positive impacts of rotations can also be delivered by increasing diversity in space, both within and among crops. Collectively, such approaches are known as polycultures, and can take a wide variety of forms, from random mixtures of two or more species (e.g. cereals and grain legume seed mixtures), undersowing of crops (e.g. cereals with clover) through to structured arrangements of intercrops (crops grown in alternate rows), boundary cropping (which might include trap crops to attract pests away from a crop, or crops grown to encourage beneficial insects for pest control), alley cropping etc.(Altieri, 1995). The range of possible arrangements is also outlined by Finckh and Wolfe (2006) on the basis of decreasing intimacy and thus interaction, from mixture to block. The choice usually involves a trade-off between gains in, for example, disease or weed control in highly intimate arrangements, against a loss in convenience of management, including mechanisation for weed control and harvesting.

Species mixtures offer numerous benefits including: improving crop growing conditions by increasing water and nutrient-use efficiency; improving soil structure and fertility; and enhancing pest, disease and weed control (Gliessman, 1995). Diverse polycultures often possess a greater ability to buffer against environment stresses, which can lead to higher land utilisation efficiency and more stable, but not necessarily lower yields (Willey, 1979; Agegnehu *et al.*, 2006). (The yield benefit is typically measured by means of Land Equivalent Ratio, which compares the total production from a polyculture with the production if each component had been grown separately – see Chapter 4.2 for further consideration of this issue.)

The benefits of polycultures relate to the enhancement of ecological function through processes such as facilitation (enabling improved growth conditions for another component), complementation (complementary resource demands of different components) and compensation (when adverse conditions inhibit one component and another is able to take its place), resulting in enhanced resilience of the system.

Facilitation is when one crop alters the growing environment of another crop leading to an increase in crop growth (Begon *et al.*, 1996). Cereals grown with grain legumes compete for nitrogen, stimulating greater nitrogen fixation by the legume than if it had been growing alone (Kontturi *et al.*, 2011). Certain crops may provide physical support for others (e.g. peas with cereals, or climbing beans with maize), or shelter from wind as in agroforestry systems. Polycultures can result in greater control of host-specific crop pests and diseases by altering microclimates and reducing spread of pathogens largely through reductions in host density (for a review, see Boudreau (2013)).

Complementation results from crop differentiation in resource requirements and usage, allowing for increased resource use efficiency and greater productivity (Vandermeer, 1992). This differentiation may occur in both space and time. For example, crops growing at different heights may complement each other in terms of access to solar radiation, while annual and perennial crops grown in combination as in agroforestry mean that resources can still be captured later in the growing season when senescence has already set in for the annual crop. Greater differentiation of resource requirements and usage within the crop reduces niche overlap between species which reduces competition intensity and results in increases in crop productivity (Andersen *et al.*, 2007). However, if deployed incorrectly, polycultures can reduce primary crop yields due to high levels of competition for resources (Akanvou *et al.*, 2007).

As well as increasing biological productivity, the diversity of crops associated with polycultures can provide a degree of risk insurance, or compensation, against failure of an individual crop, particularly for smaller producers (Lithourgidis *et al.*, 2011).

The use of polycultures tends to be more common in systems with higher labour inputs, as there can be problems mechanising some of the more complex variants, and the management of many different sowing and harvest times may be more challenging. However, structured polycultures such as intercrops, strip crops, border crops and alley crops, can be designed to accommodate machinery use and are adaptable to larger-scale, commercial systems. For some polycultures, such as grain legume/cereal mixtures (see below), the selection of appropriate varieties with similar maturation dates enables combine harvesting and subsequent mechanical separation of the crop components if required. Visual recognition software and implements mean that more complex polycultures could potentially be managed mechanically, illustrating how agroecological and technological approaches could be complementary.

Complex rotations and polycultures are not restricted to crop species. Catch crops, cover crops, green manures, trap crops, shrubs and trees can all be used to enhance diversity. Cover crops and green manures are used to facilitate the growth of the primary crop by reducing weeds, pests and diseases and/or improving soil fertility and soil structure (Lu *et al.*, 2000; Langdale *et al.*, 1991). As discussed in Section 3.2.1, cover crops including nitrogen-fixing legumes may be grown prior to the main crop such that when they decompose the nutrients mineralise and become available for the cash crop (Erenstein, 2003). By raising levels of available nutrients, legume cover crops can lead to increased yield stability without the addition of large amounts of fertiliser (Mundt, 2002), but care in management is required to avoid the risk of nitrate leaching. Non-legume cover crops, such as mustard, ryegrass or buckwheat, can also be used to capture and recycle nutrients and reduce leaching of nitrogen into the groundwater (Ranells and Wagger, 1997). Cover crops can control weeds by out-competing them for resources prior to, and during the growth of the primary crop (Teasdale, 1993).

Intercropping spring oats and tic beans

Henry Stoye who farms Eastbrook Farm on the Wiltshire Downs has experimented with intercropping for the first time this year. Spring oats and tic beans were grown together at a high seed rate, the grain was harvested together before being separated out into its components after harvest. The oats were rolled and the beans hammer-milled to create a home-grown feed for the farm's pigs. Intercropping produced higher yields than if the two crops had been grown separately. The crop also provided high levels of weed suppression. Henry was previously unable to produce a home-grown protein feed crop due to weed problems associated with growing beans as a sole crop without herbicides.

(Photo: N Fradgley/ORC)

Certain catch crops can be used as part of pest control strategies (see also Section 3.3.4), for example by encouraging the initiation of a pest's lifecycle (e.g. nematodes), but then ploughing the crop in and removing the food source before the life cycle is completed (Mojtahedi *et al.*, 1991). In another variant, wild varieties of cultivated plants can be used as trap crops to attract pests away from the commercial crop (see Section 3.3.4.3).

Inter-cropping and cover cropping are usually considered to refer only to field crops (cereals, root crops, field vegetables, temporary leys etc.). However, agroforestry also covers a wide range of polyculture that include field crops, perennials and livestock (see Section 3.6.6).

3.3.3 Weed/crop interactions and management

Weeds can be defined as non-crop plant species with negative impacts on human activities such as, but not only, food production. Even from an agroecological perspective, some plants growing within an agricultural context may appear to be completely negative in value, with docks, thistles and couch as prime candidates. However, all 'weeds' have some characteristics and effects that are positive, for example recycling nutrients, providing habitat for beneficial organisms, or providing a means of induced resistance to pests, pathogens and other weeds. These benefits are likely to be better exploited if there is a reasonable degree of weed management, rather than total removal.

Nevertheless, weeds can represent a major threat to sustainable agricultural systems and are the most important biological factor that reduces arable crop yield (Gallandt and Weiner, 2007). This is largely because weeds consume resources that would otherwise be available to the crop. Agroecological approaches to weed control include the exploitation of crops' competitive ability to suppress weeds. Crops and weeds (non-crop plants) forming plant communities where weeds typically display characteristics of early colonisers in ecological succession (e.g. high reproductive capacity and rapid germination/life-cycle progression) or high levels of adaptation to cultivation timing/methods, herbicide use and crop lifecycles (Liebman *et al.*, 2001).

In particular, the increased use of herbicides to control weeds has reduced the potential for weeds to contribute to ground cover, and is leading to the development of herbicide resistance in weeds (Wolfenbarger and Phifer, 2000; Owen and Zelaya, 2005), a process exacerbated by the advent of genetically-modified, herbicide-tolerant cultivars (particularly maize, cotton and soybean) and associated monocropping (see Figure 3-3).

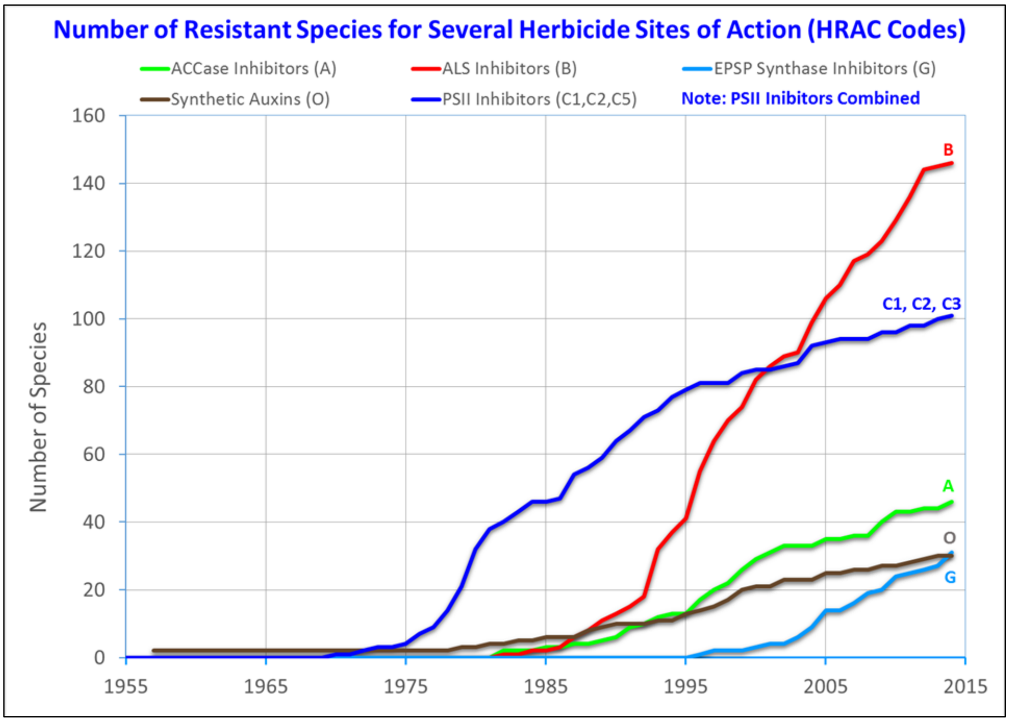


Figure 3-3: Development of weed resistance to herbicides
Source: Ian Heap, International Survey of Herbicide Resistant Weeds
www.weedscience.org/graphs/soagraph.aspx.

Competition between crops and weeds can refer to competitive effect (weed suppression) or competitive response (weed tolerance). Weed suppression occurs when a crop decreases the growth, survival or fecundity of neighbouring weeds, primarily by depleting resources such as light, water, and nutrients. This might be through a spreading growth habit, or through tall varieties of crops like cereals. Crops can also have allelopathic effects on weeds, preventing neighbouring weed germination by exuding suppressive chemicals into the soil (Worthington and Reberg-Horton, 2013), a phenomenon frequently attributed to oats. Weed tolerance involves a response to competition such as an alteration of life history, by changing the timing of specific growth periods or delaying flowering, which reduces the temporal demand for resources thereby reducing competitive intensity (Callaway, 1992).

Weed suppressive traits such as high establishment rates, rapid development of canopy cover and increased tillering ability have been shown to have negative effects on neighbouring crop plants resulting in reduced yields (Jordan, 1993; Lemerle *et al.*, 2006; Song *et al.*, 2010). When selecting different crop types (varieties, mixtures or populations) for a specific cropping system, attention needs to be paid to any potential trade-offs between reducing competition between crop plants and increasing competition against weeds. Selection should be in favour of early competitive traits and against later competitive traits, providing good weed suppression at the start of the growing season and reduced intra-crop competition later in the season during the grain filling period. These breeding targets are only likely to be achieved if selections are made in competitive environments involving high crop density and high weed pressure (Weiner *et al.*, 2001; Lemerle *et al.*, 2006). Information regarding the competitive ability of varieties is important when attempting to minimise yield losses associated with competition between crop plants and competition from weeds (Jordan, 1993; Lemerle *et al.*, 2006; Dhima *et al.*, 2010; Song *et al.*, 2010).

Breeding for crop allelopathy, and selecting species with high allelopathic potential in rotations, is another alternative approach to controlling weeds and reducing the yield penalty associated with high competitive ability. Allelopathy in crops such as oats, and to a lesser extent rye, is well documented (Kruse *et al.*, 2000). However, variation for allelopathy has been found to exist in many other important crops such as wheat, oats, barley and rice (Wu *et al.*, 1999; Bertholdsson, 2004; 2005). A lack of selection for allelopathy has resulted in a loss of allelopathic traits in many modern crop varieties, which, according to Berholdsson (2004), should be reintroduced into breeding programmes as a cost effective method of controlling weeds.

The most effective weed control strategy may involve a combination of weed suppression, to reduce weed numbers, and weed tolerance, to maintain yield stability in environments where weed competition is unavoidable (Jordan, 1993). In practice, rotation design, including alternating spring and winter sown crops, varying cultivation practices, making use of undersowing and intercropping techniques all provide opportunities to manage weed communities. Mechanical weeding practices such as ploughing, comb harrowing and interrow hoeing can also be effective at controlling weeds, but are normally dependent on fossil fuels and can cause soil degradation (Holland, 2004), so are better seen as of secondary importance to good agroecosystem design and management.

3.3.4 Pest/crop interactions and management

The increasing use of pesticides in agriculture during the 20th Century is closely linked to the increased use of industrially-fixed nitrogen, in that this enabled simplified rotations without a fertility-building break. Pest species (and pathogens) were able to take advantage of shorter breaks between host crops and plentiful food sources, including the uptake and storage of surplus nitrates by plants. As fertiliser use increased, so did pest and disease problems, stimulating the development of pesticides and fungicides to control them, often with wider impacts on non-target organisms and the environment. Carson's (1962) book, '*Silent Spring*', which highlighted the direct impacts of DDT on birds, provided a powerful stimulus, not only to popular awareness of the potential negative impacts of pesticides, but also to the development of safer pesticides and agroecological alternatives.

However, the indirect impacts of intensification may be more significant. The intensification of arable crop production in the UK between 1950 and 1990 profoundly changed the arable crop environment, with crops becoming increasingly inhospitable to a number of potentially beneficial species (Firbank *et al.*, 1992). Farmers began to grow most of their crops in the autumn rather than the spring, which created an almost year round green cover for pests and diseases to survive (Hay *et al.*, 2000). Although pesticides became increasingly selective in their mode of action, which reduced their impact on non-target organisms, their increasing use meant that beneficial organisms that might have helped control pests had reduced access to suitable food sources. Removing host plant species from field boundaries as hedges were removed and fields increased in size, as well the simplication of cropping sequences, meant fewer reserves from which beneficial organisms could colonise fields. In some cases, this generated a so-called ‘boomerang’ effect, where a pest population could rebound to greater problem levels than if there had been no direct control with pesticides, because the normal regulatory processes involving beneficial organisms were no longer functioning (Isaev *et al.*, 1994; Hawkins and Cornell, 1999).

Despite widespread awareness of these issues, there are continuing target pest resistance, environmental and health concerns with respect to pesticide use (including most recently the much-debated neonicotinoid impacts on pollinators and metaldehyde slug pellet impacts on water quality and non-target organisms). These problems raise questions about the extent to which further pesticide development and use is sustainable. However, regulatory pressures (including the EU Sustainable Use of Pesticides Directive and the risk of losing particular products), and industry initiatives responding to public concerns (such as Voluntary Initiative, the Campaign for the Farmed Environment (CFE) and Linking Environment and Farming (LEAF)), are contributing to both improved pesticide use practices and the development of more integrated approaches, many agroecological in nature.

3.3.4.1 Cultural management options

Cultural control involves the modification of cropping practices and the cropping environment as a method to reduce the prevalence of pests as well as weeds and diseases. There are a wide range of management techniques encompassed by cultural control, which include crop rotations, cover crops, intercropping, trap cropping, physical controls (e.g. nets and fleeces) and good sanitation. All of these practices act to modify the relationships between the pest species and the cropping environment, for example by creating discontinuity of food resources and/or hosts for pests and diseases.

A diverse crop rotation, formed of differing crop groups - cereals, legumes, root crops, field vegetables and broad-leaved arable crops – is a favoured method of cultural control. The aim is to create an economically successful rotation which will break many pest and disease cycles, increase weed control, reduce soil erosion and improve soil fertility and condition and minimise chemical intervention (Cook *et al.,* 2010). For oilseed rape, for example, one in three year rotations would halt the increase in club root, but a one in five year rotation would be ideal to reduce the impact on yield brought about by other soil borne pests and pathogens (Oxley, 2007; Hilton *et al.,* 2013). Studies on the control of carrot fly (*Chamaepsila rosae*) have demonstrated that a major risk factor for infestation in any one season is the distance to fields where carrots were grown in the previous year; thus implementing a rotation to ensure separation of carrot fields from year to year is a major control option (Hermann *et al.*, 2010). The continuing technical and environmental challenges of pest control and pesticide use have led conventional farmers to revisit the concept of crop rotations as a method to contribute positively to soil fertility and manage pests with reduced chemical use (Jordan and Hutcheon, 1995).

Short crop rotations can result in the build-up of soil-borne pathogens such as club root in cruciferous crops such as oilseed rape. The UK has seen a steep rise in affected oilseed rape as a result of inadequate rotations and early drill dates (Oxley, 2007). An eight-year HGCA project at a trial site in Norfolk investigated the impact of short rotations such as the more common wheat-oilseed rape-wheat rotation (Hilton *et al.,* 2013). They found clear yield penalties from shorter rotations compared with longer rotations and with crops grown on 'virgin' land where the crop had not been grown previously for both oilseed rape and winter wheat (with associated financial implications), as well as a clear negative effect of rotational intensity on volunteer numbers, crop vigour and disease levels (Figure 3-4).

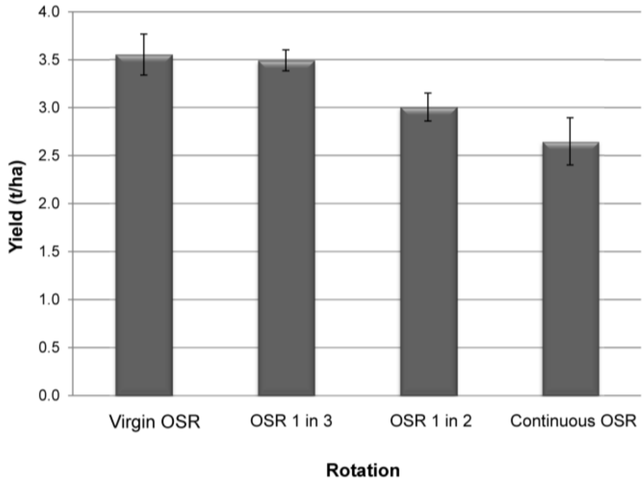


Figure 3-4: Yield data from plots within different oilseed rape/ wheat rotations from the fourth year of the field trial (2007). Error bars are ± standard errors. Source: Hilton et al. (2013).

3.3.4.2 Biological pest control, pheromone trapping and mating disruption

Particularly in horticulture, the opportunities for using rotational breaks to control pests in perennial and protected cropping are limited, and rotations are less effective against air-borne pests. Alternatives exist which can be used in isolation or in combination with one another and other control measures in an integrated pest management programme. Classical (augmentative) biological controls involve the deliberate release of beneficial organisms into the environment in order to regulate pest populations. Pheromone controls can be used in a variety of ways, for example: to attract pest insects to a trap, either in an attempt to reduce population numbers (known as lure and kill), or to monitor pest outbreaks to enable the more effective timing of other control measures; or they can be used to disrupt the mating of insect pests (known as mating disruption), during which the air is 'flooded' with synthetic pheromones, rendering the pests unable to locate one another, thereby disrupting mating and the production of offspring. These forms of control methods can often be difficult to implement and require much research and development to ensure that they function effectively in different agricultural systems.

Classical biological control often requires regular introduction of beneficial organisms to function effectively. This can be because the organisms are too effective at control, eradicating the pest and therefore leaving no food source, because they disperse away too readily, or do not have the habitat provision to survive in the agro-ecosystem longer term. Combining biological control with habitat manipulation can improve their long term efficacy. Biological control is often most effective when used in protected cropping or for control of stored product pests, where the pest and its natural enemy are enclosed within the same environment, therefore optimising their chances of encountering one another.

Combinations of different control methods, for example cultural and biological control can increase the efficacy of control measures. For example, studies in Germany have shown that covering Brussels sprouts with fine-meshed nets from the time of transplanting until the end of October achieves a 77% reduction in the infestation by cabbage whitefly (*Aleyrodes proletella*) during the main infestation period in September. Combining netting with the release of the parasitoid wasp *Encarsia tricolor* inside the nets resulted in a 39% reduction in pest density and a 23% increase in marketable yield (Schultz *et al.*, 2010). (These small insects lay their eggs inside the pest, the emerging larvae then feed off it, killing the pest).

There are risks associated with classical biological control approaches involving introduced organisms, such as the absence of appropriate hosts or ecological niches to support the introductions, and the significant risk of introducing organisms into alien environments where there may be inadequate control mechanisms to limit population growth. There is also the risk of resistance build-up, for example through the overuse of *Bacillus thuringiensis* to control pest larvae. This problem has now increased in the United States and elsewhere as a consequence of genetically-engineering the toxin produced by *B. thuringiensis* into Bt cotton and maize, with increasing evidence of the failure of the refuge strategy to reduce resistance risk (Tabashnik *et al.*, 2013).

3.3.4.3 Habitat manipulation for biological control

A common feature of agroecological systems is the creation of favourable conditions for natural (passive) biocontrol of pests through predators and parasitoids, which may be supported by a higher prevalence of non-crop species where herbicides are not used, or by the deliberate provision of refugia such as flowering field margins, conservation headlands and beetle banks. MacFadyen *et al.* (2009b) found that herbivores in organic fields are attacked by more parasitoid species, while Crowder *et al.* (2010) found that pest control was due to greater evenness of natural enemy populations, independent of species richness. Success is, however, variable because of environmental interactions with hosts and other factors (Roschewitz *et al.*, 2005; MacFadyen *et al.*, 2009a; MacFadyen *et al.*, 2009b).

Agricultural intensification has seen the removal or degradation of field boundaries such as hedgerows in favour of maximising field size (Barr *et al.,* 1991; Greaves and Marshall, 1987). Where hedges have remained, many have been ploughed up to the base and the increased use of herbicides in such close proximity has had deleterious effects upon the ground flora, favoured by many polyphagous beneficial invertebrate predators (MacLeod *et al.,* 2004). Tussocky grasses such as timothy, cocksfoot and red fescue reduce temperature variations more than other plant species providing an ideal overwintering habitat for beneficial predatory invertebrates (Thomas *et al.,* 1991). Increased field size means that it can take many weeks for beneficial non-flying predatory insects to reach the centre of a field on emerging from hibernation from field boundaries as temperatures increase in spring.

Scientists working at the Game Conservancy Trust (now GWCT) identified these problems (Potts, 1992), but recognised that farmers would be unwilling to revert to the old methods and reduced yields. Initially they developed a small number of habitat features designed to provide refuge from the hostilities of the crop production areas, including beetle banks, conservation headlands and managed field margins. Further developments arose later with research into pollen and nectar mixes designed to encourage parasitoids.

<u>Flowering field margins</u>

The presence of flowering plants in agricultural systems increases the numbers of natural enemies, including parasitoids, as well as other beneficial insects such as pollinators (Wratten *et al.*, 2012). By increasing the diversity of plant life within farmland systems, it is possible to provide refugia for natural enemies as well as alternative food sources, which many natural enemies require to survive at some point in their life cycle. Providing these resources and habitats creates a reserve for natural enemies and can enhance their effectiveness at controlling pests within the agricultural cropping systems on the farm. In a review of 24 previous studies, it was demonstrated that increasing landscape complexity through use of wildflower margins, hedgerows etc., enhanced the natural enemy populations in 74% of the studies and reduced pest pressure in 45% (Bianchi *et al.*, 2006).

Push-pull systems

Push-pull systems utilise the concept of companion planting, where two different species are planted together with the crop plant. The ‘push plant’ is intercropped within the main crop and naturally produces chemicals that are repellent to the pest that is being controlled against. The ‘pull’ plant or trap crop is planted to border the main crop and used to attract the pest so that it attacks this plant rather than the main crop. The most well-known push-pull system was developed by the International Centre of Insect Physiology and Ecology (Kenya) in close collaboration with Rothamsted Research (UK). Here, maize plants are protected from a stemborer moth by a *Desmodium uncinatum* intercrop, which emits an odour that repels the moth. The intercrop also produces chemicals from its roots which inhibit the growth of *Striga hermonthica*, a major parasitic weed in this region. The trap crop used is Napier grass (*Pennisetum purpureum*), which is very attractive to the moth. However eggs laid by the moth do not develop to maturity on the Napier grass, breaking the life cycle of the pest (Khan *et al.*, 2011). Push-pull approaches have also been developed for intensive crop production, for example the control of pea leaf weevil (*Sitona lineatus*) in beans (Vicia faba), of Colorado potato beetle (*Leptinotarsa decemlineata*) in potatoes (*Solanum tuberosum*) and of pollen beetle (*Meligethes aeneus*) in oilseed rape (*Brassica napus*) (Cook *et al.*, 2007). In the latter case, a boundary trap crop of turnip rape (*Brassica rapa*) is used, which is a preferred host for several oilseed rape pests. Similar work on the use of wild brassicas with higher levels of glucosinolates to attract flea beetles away from other brassica crops was reported by Gliessman (1995). These approaches, however, are less widely adopted than the Kenyan example because of the continuing availability of cheaper chemical controls (Cook *et al.*, *op cit.*). The increased problems with flea beetles following the withdrawal of neonicotinoids in Europe may make such strategies worth revisiting.

Beetle banks

Beetle banks are generally established through the middle of large fields providing an in-field overwintering refuge for beneficial insects, creating another source of emigration to reach the field centre more rapidly (Thomas *et al.,* 1991; 1992) – see photographs and Figure 3-5. As the tussocky grasses sown onto the beetle bank develop and mature, the protective overwintering features are enhanced through the development of thatch. Studies found polyphagous invertebrate predators in densities up to 1500 m^{-2} (Thomas *et al., op cit.*). The combination of labour costs for beetle bank establishment and the yield loss due to land taken out of production, together with the cost of the grass seed would amount to ca. £85 in the first year for a 20 ha winter wheat field. Subsequent costs would comprise gross yield loss at £30 year. However an aphid population kept below a spray threshold by enhanced natural enemy populations could save £300 per year in labour and pesticide costs for a 20 ha field; alternatively, prevention of an aphid-induced yield loss of 5% could save £660 for a 20 ha field (Thomas *et al.,* 1991).

Conservation headlands

Conservation headlands refer to an outermost margin of an arable cropped area, between two to six metres wide, which excludes herbicide and insecticide inputs, but still receives reduced fertiliser inputs. This resource provisioning has been proven to support greater numbers of invertebrates than areas receiving the main crop full spraying regime (Dover 1997; Hassall *et al.,* 1992) and both birds and small mammals are known to actively seek these areas due to the increased foraging opportunities (Tew, 1992; Rands and Sotherton, 1986; Vickery *et al.,* 2002). Invertebrates include beneficial polyphagous predatory insects, and pollinating insects such as butterflies (White and Hassall, 1994; Hassall *et al.,* 1992; Dover, 1997). The presence of some nationally scarce and declining arable flora in conservation headlands has also been positively associated with restricted inputs (Hodkinson *et al.,* 1997; Eggenschwiler *et al.,* 2007) and this enhanced vegetation structural complexity has been positively correlated with the numbers of beneficial predatory spiders found in such habitats (White and Hassall, 1994).

Conservation headlands are often ideally located in areas of least crop yields so their installation can negate the loss of productive land, such as to meet the requirements of a riparian buffer zone, enhance provision of the preferred insect food for game birds to increase farm shoot revenues, and can attract payments through agri-environment schemes. Conservation headlands not only have a value to wildlife conservation, but make a valuable contribution to the pool of predatory beneficial fauna available to the farmer, with the potential additional economic advantage of less pesticide required in the main cropped area.

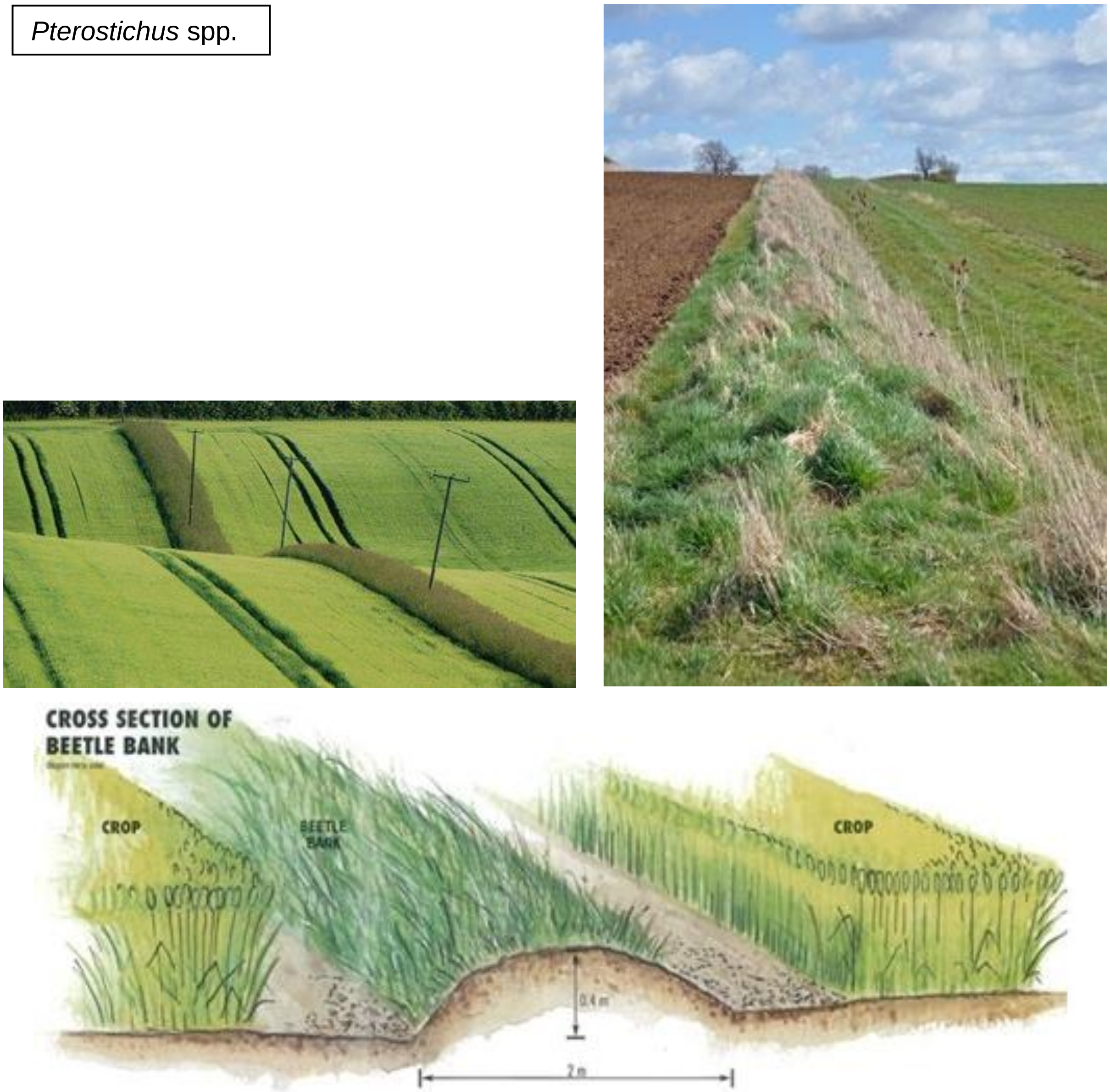


Figure 3-5: Illustrations and cross section of beetle bank. (Source: GWCT)

The benefits of the various strategies referred to in this section have also been documented by Boatman *et al.* (2010) in their review of Environmental Stewardship benefits to agriculture, highlighting the two way flow of benefits between biodiversity protection and food production. However, the pest control benefits of natural pest predation through provision of habitat have to be set against the reduction in output from taking land out of production, with the trade-offs not always well understood. There is a need for a better understanding of the optimal distribution of these habitats in the landscape. Research is being carried out on the development of pest suppressive landscapes, in particular under the EU-funded PURE project (http://www.pure-ipm.eu/). Overall, further work is needed to give farmers confidence to change their crop protection practices in this direction.

3.3.4.4 Perennial crops and agroforestry

Reduced pest problems in agroforestry systems are predicted and can be observed due to greater niche diversity and complexity than in monocropping systems (Stamps and Linit, 1998). Agroforestry systems can be managed to enhance pest regulation, for example by providing sources of adult parasitoid food (e.g. flowers), and sites for mating, oviposition and resting sites (Young, 1997; Stamps and Linit, *op cit.*). An example of this is the use of flowering understoreys in orchards (see section 3.6.6). Trees provide greater structural and microclimate diversity, greater temporal stability, greater biomass and surface area, alternative sources of pollen, nectar and prey, alternative hosts and stable refugia. Trees, hedgerows and other permanent non-cropped areas of agroecosystems provide shelter for overwintering natural enemies, as well as alternative food sources when crop pest populations are reduced following harvest (Dix *et al.*, 1995; Schmidt and Tscharntke, 2005).

However, some pest groups such as slugs have been observed in higher numbers in agroforestry systems, and shifts in the relative importance of pest groups may present novel management problems and influence crop choice. Griffiths *et al.* (1998) observed increased slug populations in agroforestry plots compared to arable controls in a silvoarable experiment in West Yorkshire. Levels of slug damage correlated with slug abundance, with lower numbers of emerging pea plants and higher levels of leaf damage in drill rows next to the tree rows than in the arable control. It was suggested that silvoarable systems can enhance slug populations and activity in two ways. Firstly, slug populations in arable areas are reduced by soil cultivations; permanent, unploughed vegetated areas under the tree rows in agroforestry systems provide refugia for both slugs and their natural enemies. Secondly, the microclimate of the agroforestry system is modified by the presence of the trees and understorey vegetation, with higher levels of soil moisture favouring slug populations (Griffiths *et al.*, *op cit.*).

An alley-cropping system with peas (*Pisum sativa*) and four tree species (*Juglans*, *Platanus*, *Fraxinus* and *Prunus*) in Leeds supported higher insect diversities and natural enemy abundance, and lower abundances of pea and bean weevils (*Sitona* spp.) and pea midge (*Contarinia pisi*) compared to a monoculture of peas (Peng *et al.*, 1993). In this same silvoarable system, grain aphid (*Sitobion avenae*) populations in the winter barley crop were approximately half that of the arable control (Naeem *et al.*, 1994). This was attributed to an increase in cereal aphid predators, primarily hoverflies (*Diptera*: *Syrphidae*), which used the tree-strips as a refuge (Phillips *et al.*, 1994).

Agroforestry systems have also been shown to support higher bird populations which are likely to contribute to invertebrate pest regulation (Williams *et al.*, 1997). In Iowa and Illinois, USA, Best *et al.* (1990) recorded seven times as many birds and twice as many breeding bird species in woody edge habitats compared to herbaceous edge habitats.

3.3.5 Plant disease management

3.3.5.1 Soil-borne diseases

Soil organic matter content and soil microbial activity can influence the levels of soil-borne pathogens and plant resistance to them, with strong evidence that organic soil amendments such as compost can enhance soil pathogen suppression (Cook and Baker, 1983; Hoitink and Boehm, 1999; Bruggen and Semenov, 2000). It is likely that organic matter provides nutrients and energy to support diverse soil microbial communities that compete with pathogenic organisms and inhibit their development. Amendments such as compost also contain high levels of microbes that may enhance this diversity. Crops grown in soils with high organic matter content and diverse and active microorganism communities generally show higher tolerance or resistance to diseases (Altieri and Nicholls, 2003). Agroecological approaches such as organic and low-input farming can increase levels of soil organic matter and enhance soil microbial characteristics (Ge *et al.*, 2011). Higher levels of soil organic matter in agroforestry systems also positively influence soil invertebrate communities (Park *et al.*, 1994; Price and Gordon, 1999). In a poplar-arable rotation silvoarable system, soil organic matter, soil arthropod abundance and cumulative body mass were higher in samples taken close to the trees, with lower levels in the crop alleys attributed to frequent cultivations, lower litter inputs and a reduction in tree root densities (Park *et al.*, *op cit.*).

High inputs of synthetic/inorganic fertilisers can lead to nutrient imbalances, in particular the surplus uptake of nitrogen as nitrate-N, which can be stored by the plant and provide a nutrient reserve for pathogens; which is one of the reasons why fungicide use may increase with increased fertiliser use. Where nitrogen is primarily derived from organic sources, a higher proportion is taken up by the plant as ammonium-N in the early stages of nitrification (see Figure 3-1 above), reducing the potential for pathogen infestations (Huber and Watson, 1974).

Equally, low levels of fertilisation can result in disease problems due to nutrient deficiencies. This may be reduced where soil microbes help compensate for low soil nutrient availability. For example, arbuscular mycorrhiza (AM) fungi symbioses have been found to reduce infections with soil-borne diseases (Azcón-Aguilar and Barea, 1997; Jeffries *et al.*, 2003). This protective effect was shown to be dependent on the specific pathogen involved, and some AM species and isolates were more effective than others in protecting the plant from infection. Jeffries *et al.* (*op cit.*) highlight AM fungi as being the 'most important microbial symbioses' for most plant species, showing their greatest effect in soils with lower nutrient contents, particularly P-limitation.

Some important soil-borne cereal pathogens occur at considerably lower levels in organic compared with non-organic systems. This is partly due to lower cereal frequencies in the rotation, the reduced use of winter varieties, and the fertility building break in organic rotations providing an opportunity to avoid green bridges and break pathogen lifecycles. However, another factor is the difference in frequency of fungal and bacterial competitors of the pathogen. For example, Hiddink *et al.* (2005) showed that organically managed soils were better at supporting the bacterial antagonist, *Pseudomonas fluorescens,* of the take-all pathogen *Gauemannomyces graminis*. *P. fluorescens* is a well-known example of plant growth-promoting rhizobacteria (PGPR) which, together with other soil-borne organisms, are able to induce resistance against pathogens of both the roots and upper parts of crop plants and also plant pathogenic nematodes (see also, Shennan (2008)).

The use of specific crops or green manures for the reduction or suppression of specific diseases and plant pathogens (biofumigation) has proved successful in trials. For example, a reduction of *Rhizoctonia* diseases was found in potatoes grown following rapeseed and canola green manures (Angus *et al.*, 1994; Larkin and Griffin, 2007) and following mustard green manures (Sexton *et al.*, 2007). Brassicas in general, particularly those showing high glucosinolate contents (such as 'Caliente', white- and oriental mustards) have been found to be very effective in the suppression of pathogens (Sarwar *et al.*, 1998). In combination with other integrated and agroecological practices, the use of green manures for biofumigation can increase crop productivity, as well as the efficiency and sustainability of disease management systems (Larkin *et al.*, 2012).

3.3.5.2 Air-borne diseases

Air-borne diseases can be reduced through the application of within-species diversity such as the use of varietal mixtures, in which multiple varieties are grown together in the same field. Varietal mixtures can control disease in several ways, including the prevention of pathogen spread by increasing the distance between susceptible host plants, or the use of resistant plants to form a barrier to prevent pathogen dispersal (Chin and Wolfe, 1984; Zhu *et al.,* 2000). The beneficial effect of mixtures on disease control has been observed in many crops, controlling major pathogens such as powdery mildew in barley (Wolfe 1992) and stripe rust in wheat (Finckh and Mundt, 1992).

The concept of management of rhizosphere ecology (suppressive soils) to control soil-borne diseases can be extended to control of leaf-surface pathogens using microbially-rich extracts from compost and other sources (Weltzien, 1991; Bailey *et al.*, 2006).

The potential for agroforestry to reduce disease pressure in trees has not been investigated fully, but it is likely that widely-spaced trees could be less susceptible to some tree diseases. A current EU-funded project, CO-Free[15], is investigating the potential of agroforestry as a strategy to replace copper-based products as plant protection products in organic top fruit systems. Integrating top fruit production into an agroforestry system, where woody species are integrated with crop production may have a beneficial effect on the control of plant pathogens such as scab (*Venturia inaequalis*) due to a number of mechanisms:

- a greater distance between tree rows in agroforestry systems, with crops in the adjoining alleys, is likely to reduce the spread of pathogens - this has been recorded for other crop pathogens (Schroth *et al.*, 1995);
- lower densities of trees favour increased air circulation which has been shown to reduce the severity of scab by reducing leaf wetness duration (Carisse and Dewdney, 2002);
- regular cultivations within the crop alleys will incorporate leaf litter into the soil, thus enhancing decomposition and reducing the risk of re-inoculation from winter-surviving scabbed leaf litter the following Spring.

Preliminary data from 2012 (Smith, unpublished) indicated that scab levels were less than half in the organic agroforestry site (WAF) compared to the organic orchard control (CLO) (Figure 3-6). Neither system used copper or any other inputs.

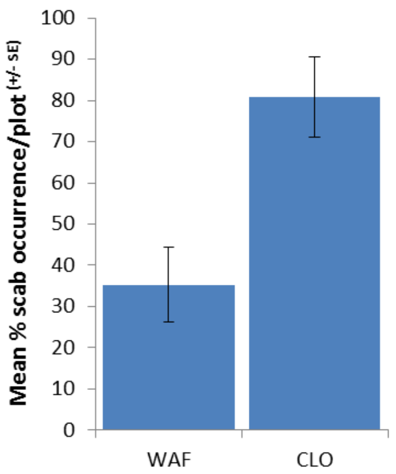


Figure 3-6: Mean scab occurrence (± SE) on pre-harvest apples in an organic agroforestry system and control orchard in 2012.
Source: Smith (unpublished)

3.4 Animal production systems

Although not the primary focus of much of the agroecology literature[16], animal production, health and welfare can all be influenced by agroecological practices and system components, both in the context of species composition of grassland (including legumes, herbs and woody perennials) and in terms of stocking density, animal species interactions and grazing management practices. In some cases agroecological systems specifically

[15] http://www.co-free.eu/
[16] The journal *Animal* published a special issue on agroecology in 2014 (Vol 8(8)). The editorial claims that the recent surge in literature on agroecology has ignored animal production systems.

targeting livestock production have been developed. Although we have not covered housing design specifically in this review, agroecological approaches normally exclude permanent housing of livestock, including the use of feedlots, as such approaches significantly restrict the interaction of animals with the agroecosystem, and the production, health and welfare benefits that can be derived from such interactions.

3.4.1 Integration of livestock and crop production

For many, mixed farming (integrating crop and livestock production) is seen as an important element of agroecological systems, with an emphasis on closing nutrient cycles and meeting the animal's nutritional requirements from within the farm system. Animals can utilise the herbage legumes and grasses that form the basis of the fertility-building phase of arable rotations, as well as other by-products of crop production such as straw. The nutrients and energy recycled through their manures can be used to support soil fertility and crop production. While this integration is often conceived of as taking place on individual holdings, it can also result from co-operation between holdings, as is the case with the GWCT Allerton Project (Leake, pers.comm.).

The EU-funded project CANtogether[17] is currently revisiting the question of how crop and livestock production can be better integrated to optimise farming systems. Moraine *et al.* (2014) describe the participatory process being used to design integrated crop–livestock systems (ICLS) as part of the ecological intensification of agriculture. Using both 'metabolic' and 'ecosystemic' approaches, they argue that a diversity of crops and grasslands interacting with animals is central for designing systems that provide and benefit from ecosystem services. Coquil *et al.* (2014)[18] examined the potential for mixed crop-dairy systems, and the factors which influence producers adopting them. This included the introduction of temporary leys in crop rotations and rotational grazing, as well as other strategies to reduce costs and become more self-reliant/autonomous in decision-making.

The integration of animals can often make the difference in realising long-term ecological sustainability and socio-economic viability (Dumont and Bernules, 2014). In the following sections we have examined the contributions of agroecology to grassland ecology and management, animal nutrition, and animal health and welfare, for both ruminant and monogastric livestock species.

3.4.2 Grassland ecology and management

Permanent grassland accounts for 30% of EU agriculture in general, but as much as 47% of the total organically-managed land area (Huyghe *et al.*, 2014). This reflects the tendency for a higher proportion of extensive grassland farms to adopt organic methods as well as the increased importance of grassland in organic and other agroecological systems. Grasslands and their management play an important role in the nutrition of animals, often permitting production in regions not suitable for the production of crops directly for human consumption. They also impact on the health of livestock, including metabolic diseases and parasites, and support a wider range of biodiversity including key pollinators (Brown, 2014).

According to Younie (2012), the organic grassland farmer faces the dilemma whether his first priority should be avoiding parasite build-up in the sward and herd or to maximise grazing efficiency. Under an agroecological approach, grassland management strategies should focus on a range of outcomes which include: achieving or maintaining a high animal health status; an appropriate botanical composition of the swards with high biodiversity; and good soil fertility, carbon storage and water quality. These are as important as maximising

---

[17] http://www.fp7cantogether.eu/index.php

[18] Several French studies on Integrated Crop-Livestock systems have been published as a special issue of *Renewable Agriculture and Food Systems* (Vol. 29(3), September 2014).

herbage output per hectare in the short term (Huyghe *et al.*, 2014). However, livestock production in Europe, including on many organic farms, at present relies on relatively few herbage species.

A more agroecological approach would be to establish diverse swards with high nutritive and medicinal value for animal productivity and health. By using mixtures of functionally diverse plant species, synergies between agricultural productivity and other ecosystem services can be optimised and fine-tuned to farm-specific needs (as has been seen within the Defra-funded LegLINK project (Döring *et al.,* 2013)). Secondary grassland species, including herbs and legumes, could help optimise the productivity of low fertility areas, as such species often require less nutrients and will thrive under sub-optimal soil conditions.

Research in sown grasslands has shown that increasing the number of plant species in established grass swards offers considerable potential to support ecosystem services including biomass production (Tilman *et al.*, 2006; Weigelt *et al.*, 2009). Due to the artificial character of such experimental grasslands, it has been questioned how far the results would also apply to agriculturally-managed grasslands (Wrage *et al.*, 2011). European research with intensively managed leys revealed the diversity benefits for biomass production and weed suppression of seed mixtures of grasses and legumes with between one and four dominant and highly valuable forage species (Kirwan *et al.*, 2007; Lüscher *et al.*, 2008). Suter *et al.* (2015) provide the most recent evidence from these studies, demonstrating that species diversity in grassland can increase both nitrogen and biomass yield compared with the same species grown separately, similar to the evidence for polycultures and agroforestry discussed elsewhere in this report. It is likely that there are not only complementary demands for resources between the species, but that the grass species competing for nitrogen stimulate increased biological nitrogen fixation by the legume component.

Research on a variety of secondary grassland species also indicates potential to support grass and ruminant production through sustained yield and complementary herbage quality (Frame, 1991; Bruinenberg *et al.*, 2002; Lange *et al*., 2011). Such species may occupy niches in a grassland system and exploit above and below ground resources that would otherwise not have been utilised, thereby stabilising production. Increased diversity is also expected to be beneficial to the welfare of ruminants since they can use their exploration and choice capacities during grazing which has impact on animal nutrition (see Section 3.4.3).

Management by grazing is essential for the maintenance of structure, balance, diversity and longevity of species-rich grasslands (Fleischer *et al*., 2013). Grazing animals can produce benefits to sward structure and development through recycling of nutrients and other factors. However, there is limited evidence on the effects of the duration of the resting period during rotational grazing on productivity and persistence of diverse swards. This is relevant in the context of recent interest in the concept of mob-stocking, which involves long resting periods followed by short grazing periods with high stocking rates, with much of the vegetation trampled underfoot contributing, it is argued, to the build-up of soil organic matter levels. This approach has been popularised by Savory (Savory and Butterfield, 1999), although the scientific evidence and the applicability of this approach, both in the UK (under higher rainfall conditions than in Africa and the United States, where the approach was first developed) and elsewhere, has been questioned (Measures, 2014; Carter *et al.*, 2014; Savory, 2013).

According to McCarthy *et al.* (2014), the nutritional management of grazing cattle is heavily influenced by stocking rates. As stocking rate increases, grazing animals are less likely to achieve their dry matter intake potential which can affect animal productivity. If the stocking rate is excessively low, pasture is wasted and sward quality deteriorates. Moreover, the availability and nutritive value of grazed grass varies over the grazing season. As plants mature, pasture growth rates are high but digestibility declines (McCarthy *et al.*, *op cit.*). The challenge is to identify optimum stocking rates over the grazing season so that dense palatable swards are maintained and good productivity per hectare is achieved without compromising animal health and welfare.

Optimising dairy performance through the use of diverse swards

Rob Richmond has made good use of diverse swards on his 225 ha dairy farm in Gloucestershire. Herbal leys are maintained for up to seven years and consist of a diverse mix of species such as trefoil, chicory and plantain. The high fibre and micronutrient content helps to improve herd health while the improved palatability increases voluntary feed intake. Mr Richmond states that a change in farmer attitudes is required with regard to diverse swards, as is it necessary to 'get over it looking untidy' in comparison to simple ryegrass leys. In his view, the improvement he has observed has more than compensated for the unconventional appearance.

(Photo: ORC)
(From: www.cotswoldseeds.com/seed-info/case-study-creating-grazing-mix-when-ryegrass-not-option)

3.4.3 Animal nutrition

Grassland and particularly pasture-based feeding systems with low concentrate supplementation can achieve similar animal performance to indoor systems with higher dietary concentrate proportions, without negative impacts on metabolic health (Rauch *et al.*, 2012; Ertl *et al.*, 2013).

In agroecological monogastric systems, the animals would ideally derive part of their nutritional requirements from rangeland resources, i.e. plants and invertebrates. In practice, however, it is very difficult to account for the contribution of these resources to meeting the needs of the animals and they may vary depending on the stage of production or the availability of certain nutrients in the rest of the diet. For instance, ration planning in most organic monogastric production, like conventional systems, relies primarily on feeds containing cereals and oilseed products (Edwards, 2003), with limited recognition given to the nutritional potential of the range. This can result in high nutrient losses from these outdoor systems as only a proportion of feed N input is retained by the animal (e.g. 30% feed input retained in pigs until slaughter) (Eriksen *et al.*, 2006), leading to concerns regarding nitrate leaching and eutrophication. Recent studies have indicated that reducing the input of supplementary feed can encourage foraging. In these cases, the animals are capable of finding and utilising considerable amounts of different foods from the range to balance their diets without negative effects on welfare or productivity. For example, Horsted (2006) found that foraging on a diverse range area with abundant vegetation can contribute significantly to the nutritional needs of high producing laying hens. They estimate that nutrient-restricted hens in some periods had up to 70% of the lysine and methionine requirement provided through the forage material (Horsted, 2006; Horsted and Hermansen, 2007).

Similarly, it is well documented that herbage intake has the potential to make an important contribution to mineral, trace element and vitamin supply for pigs, for example meeting 50% of the maintenance energy requirement and a high proportion of the amino acid, mineral and trace element requirements of dry sows (Edwards, 2003). In German experiments on the use of roughage in outdoor rearing of pigs, it was shown that Jerusalem artichoke can lead to significantly higher weight gains, compared to a standard concentrate-based ration, while weight gain decreased significantly in some cases using other roughage (Sappok *et al.*, 2008).

While it is possible to modify the vegetation (and thus the available nutrients from forage) within the range through seeding and management, the associated fauna are to a large extent an unknown quantity. Chickens have been reported to feed on a wide range of invertebrates living in the surface soil including ground beetles (*Carabidae*), rove beetles (*Staphylinidae*), spiders (*Araneae*) and earthworms (*Lumbricidae*). Pigs have evolved as opportunistic omnivores that forage above as well as below ground. When kept in semi-natural environments, they eat a wide range of items including invertebrates (Andresen, 2000; Edwards, 2003; Jakobsen, 2014). Studies reported in Jakobsen (2014) have recorded 300 earthworms in the stomach of a single pig, and an intake of 414 to 1224 worms per day by village pigs weighing 20-40 kg. For monogastric species, animal proteins are of higher nutritive value than those from plant origins with respect to particular amino acids such as lysein and methionine (Ravindren and Blair, 1993). Insects have a high nutritive value; the protein content of edible insects ranges from 30% for wood worms to 80% for certain wasp species (Khusro *et al.*, 2012). Similarly, earthworms can also contribute significantly to meeting protein requirements, with crude protein content reported as 610 g kg^{-1} dry matter for *Eisenia foetida* (Bassler *et al.*, 2000) and a mean of 43.8 and 9.2 mg gDM^{-1} lysine and methionine respectively for different species (Pokarzhevskii *et al.*, 1997). Meal from cultivated invertebrates such as house fly larvae and pupae, earthworms, silkworm pupae, grasshoppers, bees and crickets have been used in animal feed (ADAS, 2005). Such systems are currently limited by the costs of production, which means that invertebrate proteins are more expensive to produce than plant proteins. However, this would be different if the animal protein is consumed *in situ* and does not require further processing. In some habitats, naturally-occurring invertebrate densities can be quite considerable, e.g. 322-480 earthworms m^{-2} in clover grass fields, equating to a total biomass availability of 82-135 g m^{-2} (Eekeren *et al.*, 2010), and so may be able to contribute to the diet of foraging monogastrics.

Returning to plant-based feed resources in agroecological systems, browsing from trees and shrubs plays an important role in feeding ruminants in many parts of the world, particularly in the tropics. There has been considerable research into the nutritional potential and limitations of many tropical fodder species (Devendra, 1992). However, comparatively little is known about the potential of temperate browse species. Traditionally, many species of deciduous trees have been used for fodder, in particular Wych or Scots Elm (*Ulmus glabra*), ash (*Fraxinus excelsior*), silver birch (*Betula pendula*), downy birch (*Betula pubescens*) and goat willow (*Salix caprea*) (Austad and Hauge, 2006). In Norway, cattle and pigs were primarily fed leaves of *Ulmus glabra* and *Fraxinus excelsior* while leaves of *Betula s*p. and *Alnus* sp. were given to sheep and goats (Austad and Hauge, *op cit.*). More recently, the productivity and nutritional value of novel species such as black locust (*Robinia pseudoacacia*), Tagasaste or tree lucerne (*Chamaecytisus palmensis*), and thornless honeylocust (*Gleditsia triacanthos*) have been the subject of investigation, particularly in silvopastoral systems of North America and the Mediterranean (Barrett *et al.*, 1990; Burner *et al.*, 2005; Burner *et al.*, 2008; Papanastasis *et al.*, 2008).

3.4.4 Animal health and welfare

While grazing diverse swards has potential nutritive value contributing to animal health, grazing and pastures are also a major risk factor as they are the medium for development of gastro-intestinal nematodes. Ingested forage is the main source of nematode infections for grazing ruminants, but it can also affect the animals' response to infection as several plants have anthelmintic properties. Grazing management strategies can reduce the risk. Legumes and herbs offer a new perspective in the strategy to achieve lower medication use on farm level. It is well known that certain legumes and forage herbs contain plant secondary metabolites (condensed tannins, sesquiterpene lactones) that can slowdown the dynamics of gastro-intestinal nematode (round worm parasite) infections (Wink, 2012), in part by protecting proteins from digestion in the rumen and increasing mineral content, both enhancing immune system response (Marley *et al.*, 2006). Condensed tannins can bind to dietary protein, resulting in reduced protein degradation in the rumen, providing potentially increased intestinal protein supply for the grazed animal (Lorenz *et al.*, 2014). However, it is less well known how best to harness the benefits of these plants predictably and routinely under a wide range of management strategies, such as ley/arable rotation, grazing, mowing (for hay or silage) and specific soil and climatic conditions and how this impacts on overall farm performance.

The eggs of gastro-intestinal nematodes are shed in faeces. They develop into infective larvae that can be extracted from herbage and counted (Gruner and Cabaret, 1985). Pasture infectivity (indicating the risk of infection for grazers) can be expressed as the number of infective larvae per square metre or per kilogramme of herbage. The success of the development of eggs into infective larvae and the subsequent survival of these larvae is highly dependent on the micro-climate. The seasonal dynamics of infection have been extensively studied using pasture contamination studies (Gruner and Cabaret, 1985). There is evidence that forage herbs (i.e. chicory (*Chicorum intybus*) or birdsfoot trefoil (*Lotus corniculatus)* can restrict the movement of the gastro-intestinal nematode larvae in the sward, and hence the infectivity of the pasture and thus the larval uptake by stock during grazing (Marley *et al.*, 2006). There is also evidence that a high presence of chicory can influence pasture ecology, with more rapid faecal decomposition limiting the emergence and survival of the parasite in its free-living phase (Williams and Warren, 2004).

More widely used than novel forages are clean grazing systems involving either rotational interchange between livestock species, to break the lifecycles of parasites by extending the period before susceptible animals return to graze the same pastures (Keatinge, 1996). In some cases, reduced stocking densities, or the dilution of susceptible animals with either mixed species stocking or mixed age groups, may also be relevant.

Overall animal health and welfare on farms is not only the result of adopting specific practices, but a reflection of the overall management. Taking a more preventive approach to health management is not unique to organic farming but potentially more prevalent on such farms. Organic agriculture guidelines stipulate that all the preventive measures should be taken in feeding, keeping and breeding to ensure animal health and safety. It is common practice amongst UK organic control bodies to encourage producers to develop an animal health plan together with their vet. While ill animals should be treated appropriately and suffering should be prevented, the prophylactic administration of allopathic veterinary drugs is not permitted. Kilbride *et al.* (2012) concluded that enterprises participating in organic or farm assurance inspections were more likely to comply with welfare legislation in animal health inspections and that such membership could be included in the risk-based selection of farms for inspection.

Sheep grazing chicory (Photo: Anon)

Sheep find relief in the shade (Photo: J Smith/ORC)

Trees in animal production systems can also contribute to animal welfare. In addition to a diversity of foraging resources (see above), they provide shelter from rain and wind, shade from the sun and cover from predators. Cattle in both tropical and temperate climates are particularly sensitive to heat stress. Evaporative cooling is the primary mechanism by which cattle reduce their temperature, and this is affected by humidity, wind speed, and physiological factors such as respiration and sweat gland density. By providing shade, trees can reduce the energy needed for regulating body temperatures, and so result in higher feed conversion and weight gain. Research in the southern United States found that cattle that had been provided with shade reached their target weight 20 days before those with no shade (Mitlohner *et al.*, 2001). Higher respiration rates and lower activity rates of those cattle without shade were thought to reduce productivity. Evidence of benefits of shade for lactating dairy cows in temperate climates has also been found (Kendall *et al.*, 2006).

During cooler months, windbreaks and shelterbelts provide valuable protection from the wind for livestock, particularly for new-born lambs and freshly shorn sheep. When livestock have been protected from winter storms by windbreaks, significant savings in feed costs and improved survival and milk production have been reported by producers in Dakota, US (Brandle *et al.*, 2004). European research on the benefits of shade and shelter for the animal appears more limited (see for example Sibbald, 2006). One Finnish study observing behaviour at calving indicates some preference for using places that do provide shelter (Lidfors *et al*., 1994).

Ranging behaviour in chickens is affected by the type of outdoor environment provided. Dawkins *et al.* (2003) observed ranging behaviour in commercial free-range broiler systems and recorded a maximum of only 15% of the total flock outside the house at any one time. The number of birds ranging outside was correlated with the percentage tree cover on the range, and behavioural studies showed that trees and bushes were the preferred habitat (Dawkins *et al.*, *op cit.*). Descended from the forest-dwelling red junglefowl (*Gallans gallans*) of India, China and south-east Asia, it is unsurprising that chickens prefer to range in tree and thicket cover. Trees offer protection from aerial predators in particular, and can provide an escape from aggressive behaviour within the flock as well as reducing visual stimulation that can provoke aggression (Yates *et al.*, 2007). The trees also benefit from the interaction with the animal and higher leaf nitrogen concentrations and increased total height was recorded for three-year-old black walnut trees (*Juglans nigra*) fertilised with a chicken manure compared to a non-fertiliser control (Ponder *et al.*, 2005).

In the UK, all laying-hen producers within the McDonald’s Restaurants Ltd egg-supply base are required to plant at least 5% of the range area with trees (Bright *et al.*, 2011; Bright and Joret, 2012). Multiple retailers such as Sainsbury’s also promote Woodland Egg initiative in partnership with the Woodland Trust. Research has shown that the tree cover has benefits for animal welfare. Plumage damage, a key animal-welfare indicator for laying hens in non-cage systems, was found to be negatively correlated with the percentage of canopy cover within tree-planted areas (Bright *et al.*, *op cit.*). In another study of the same producers, researchers investigated whether there was a difference between two production traits – egg seconds (grade-out at the packing station indicating shell quality which is influenced by nutrition, stress and bird health) and mortality – in matched free-range laying flocks with and without tree cover on the range (Bright and Joret, *op cit.*). They found that in flocks with tree cover, there were fewer total egg seconds and significantly fewer ≥45 week egg seconds (when egg seconds are a particular problem) and lower mortality ($p=0.1$) than in flocks without tree cover.

Like chickens, pigs have a forest-dwelling ancestor, the Eurasian wild boar (*Sus scrofa)* which is found primarily in mixed, predominantly deciduous woodland. Behavioural studies of domestic pigs have shown that trees encourage expression of normal behavioural patterns (Stolba and Woodgush, 1989). Domestic pigs are particularly susceptible to heat stress, heat stroke, porcine stress syndrome and even death at temperatures above 22°C, and can suffer from sunburn and dermatitis in direct sunlight (Brownlow, 1994). Conversely, low temperatures increase the prevalence and transmission of disease. Reproductive success of domestic pigs is also influenced by temperature, with a reduction in live litter sizes with decreasing temperatures, and reduced interactions between sows and boars in poor weather lowering fertility (Brownlow, *op cit.*).

3.5 Human interactions

While it is common to perceive farming systems primarily in terms of their non-human components, social scientists recognise that agriculture is a human activity system, where humans control and interact with the system as well as the external environment. Many agroecologists argue that these social components, at both farm level and in terms of interactions with food system actors, consumers and society at large, are part of the agroecosystem. As Wibbelmann *et al.* (2013) describe, these components include issues of equity, trade, access to land, employment, working conditions and education as well supply chain, food sovereignty and consumption issues. In the UK and elsewhere in the EU, regulatory protections, for example with respect to education, employment rights and health and safety, address many but not all of the issues that agroecologists working in other parts of the world identify as key concerns. These concerns are not restricted to agroecologists. In the Square Meal report (Anon, 2014), several British research and other organisations[19] argue for the need to build a more sustainable and resilient food system by linking food, farming, health and nature, promoting local food economies and reconnecting people with their food and with producers.

Of particular relevance in the UK context is the use of knowledge and management (or design) to complement or replace some technological inputs (see also Section 2.2), which is clearly relevant to priorities for sustainable intensification (Buckwell *et al.*, 2014). Part of this reflects the various types of knowledge that support agriculture, including some that are not necessarily part of agricultural extension and education at present. Another element is to aim for closer links with consumers. These aspects are explored in more detail below.

Many writers, as in the Square Meal report (Anon, 2014), argue that improving the sustainability of the food system also requires changes to the human diet (see for example Garnett *et al.* (2013)). This is particularly relevant in the context of food security and arguments about the need to increase production to meet current projections of human food demand, including unabated increases in consumption of livestock products (see Section 4.2 on productivity). In developing country situations, as in much of Africa, agroecological approaches may contribute to increases in production by supporting the self-reliance and autonomy of resource-poor farmers (Scialabba and Hattam, 2002; Scialabba *et al.*, 2014). Pretty *et al.* (2011) concluded that sustainable intensification projects could potentially have benefits in terms of rural development as well as the environment, by fostering new social infrastructure and cultural relations, helping the emergence of new businesses and so driving local economic growth, ultimately improving the well-being of both rural and urban populations. However, not all projects that would consider themselves to contribute to sustainable intensification necessarily provide these benefits. These issues are closely related to the question of governance of food systems and the linked debates about food sovereignty, whereby production has social, cultural and economic impacts that go far beyond the direct impacts on the environment (Garnett and Godfray, 2012). However, these issues are not elaborated further here because they are outside the scope of this report.

---

[19] The Food Research Collaboration consists of the RSPB, Friends of the Earth, the National Trust, the Food Ethics Council, Sustain, the Wildlife Trusts, the Soil Association, Eating Better and Compassion in World Farming. See: http://foodresearch.org.uk/square-meal/

3.5.1 Knowledge exchange and management to foster innovation

Systems that focus on reduced input use are commonly said to do so by making greater use of information and ecological understanding (Lockeretz, 1991). Knowledge and information are therefore important components of agroecology and sustainable intensification (Buckwell *et al.*, 2014). According to Titonell (2014), knowledge sharing between farmers is at the heart of the agroecological movement. Similarly, Pretty *et al.* (2011) state that sustainable agricultural systems make productive use of human and social capital in the form of knowledge and capacity to adapt and innovate to resolve common landscape-scale problems. Based on experience in Africa, the authors emphasise how important it is that farmers see for themselves that the increased efforts and added complexity can result in substantial net benefits to productivity, but they also need to be assured that increasing production does lead to increases in profit (Pretty *et al.*, 2011).

There is a need to examine what *types* of knowledge are considered important for farming. Garforth (2010) concluded that the understanding of farmers' knowledge needs has matured from a technology transfer perspective. He identified five main areas, of which the first appears to be particularly important in the context of agroecology's contribution to sustainable intensification:

1. an understanding of the basic systems that sustain food production;
2. an understanding of the impact of current/future technologies in a specific context;
3. business management information;
4. information on markets;
5. information about the regulatory and policy context.

There has also been a focus on knowledge systems in agriculture and the role of knowledge in supporting a move towards sustainable agriculture at the EU level. For example, the European Commission's Scientific Committee for Agricultural Research (SCAR) initiated an expert working group on Agricultural Knowledge and Innovation Systems (AKIS) (SCAR, 2012). One key conclusion in the final report is that current knowledge systems in Europe are not necessarily fit for purpose to help farmers address the challenges they face, and in particular to guide them on how to implement sustainable intensification in practice. Coherent policy strategies to support agricultural knowledge systems in achieving this aim are also scarce (SCAR, 2011).

Similarly, Buckwell *et al.* (2014) conclude that the sustainable intensification of agriculture must focus on increased knowledge of how physical inputs can be combined and managed. The authors conclude that as part of sustainable intensification of European agriculture, the input of knowledge per hectare has to be intensified. This includes "*knowledge about how to manage the ecosystem services on which agriculture relies [and] to produce food outputs with minimal disturbance to the natural environment, and more environmental outputs too."* (Buckwell *et al.*, *op cit.*, p 8).

A theme running through Buckwell *et al.*'s report is that farmers (and those working with them) need accepted definitions, measurements and indicators of the state of resources and sustainability so that they can judge for themselves how well they are performing. Blum, in a case study of European land quality (Chapter 6 of Buckwell *et al.*, *op cit.*), argues that achieving action towards increasing environmental sustainability of soil management is highly dependent on having sound measurements of the underlying conditions of Europe's soils. Hill (2014) also sees effective evaluation and monitoring (broad and long-term, as well as specific and short-term) as one of the key mechanisms for achieving greater sustainability in agriculture.

The term 'agroecology' implies that knowledge about the ecology of agriculture is an essential component of the agricultural knowledge system. It is also important to identify the role farmers play in the knowledge system. Garforth (2010) cites Farming and Wildlife Advisory Group (FWAG) as one successful example that is related to agroecology. FWAG aims to encourage more biodiversity on UK farms through actively supporting farmers (see also Cox *et al.* 1991, cited by Garforth (2010)). FWAG recognised and worked with farmers as active *seekers* of advice and information, creating opportunities for them to learn how to improve conservation within productive systems that sustain livelihoods, rather than seeing them as passive recipients of knowledge dissemination.

The French/UK farmer organisation Biodiversity, Agriculture, Sustainability, Environment (BASE) places great emphasis on the fact that farmers determine for themselves what the organisation does, which gives them independence. Bellon and Ollivier (2012, cited by Caron *et al.,* 2014) stress a '*double rooting in science and in agriculture*' of approaches to ecological intensification and refer to the empowerment role for the farmers. Similarly, Kloppenburg (1991) argued that '*ecological innovation cannot occur without an epistemic change that would put farmers back at the centre of knowledge production'*. This is not just an issue for agroecology – the case for producer engagement in knowledge generation has also been made by Burgess and Morris (2009) in the context of technological development.

These perspectives stand in stark contrast to the way Morgan and Murdoch (2000) describe the changing situation for the arable sector in England in the post-war period. They say "*Efficiency came very quickly to mean the application of the new agricultural technologies which were beginning to emerge onto the market. Input companies were investing heavily in research and technology development*". As part of developing the chemical inputs for arable production, farmers' 'know-how' was replaced by 'know-what', i.e. what input to use and when. Knowledge became more codified and farmers became recipients of standardised knowledge in the form of recommendations (blueprints).

Morgan and Murdoch (*op cit.*) argue that there is therefore a greater need and scope for local, tacit forms of agricultural knowledge. In particular, knowledge about the environmental context is often specific to the location (Buckwell *et al.*, 2014). Similarly, Curry and Kirwan (2014) conclude that the complex set of objectives, values and styles of implementing sustainable agriculture at various locations do not lend themselves to reductionism or universal knowledge alone and tacit knowledge is recognised as very important.

Farmers and local communities can no longer be viewed solely as beneficiaries, but need to be recognised as active contributors to the knowledge system. However, this is not to say that science-derived knowledge can be replaced by farmer knowledge, but rather scientific knowledge must be complemented by farmer knowledge.

To make use of agroecology in supporting sustainable intensification, two types of knowledge are important:

- *Science-based and technological knowledge.* Such knowledge can be generalised and includes knowledge of basic ecological principles and the state of resources and ecosystem services on which agriculture depends;
- *Farmer (tacit) knowledge.* Even if at present it remains hard to define what exactly this tacit knowledge involves, this is grounded in farmers' observations of the various parts of their system and of the local environment.

Adopting agroecological practices consists in some cases mainly of using and working with these different types of knowledge. Padel *et al.* (2010) argue that, in such systems, innovation in the broadest sense (including social/organisational as well as technological innovations) is generated through the application of existing knowledge, for example through developing and prototyping management practices building on the knowledge of ecological principles. Access to the know-how about the principles, models and practices that can be used to counteract certain threats, combined with observation of the state of the threat, is crucial to the farmer's ability to respond effectively to new challenges, such as conservation and protection of natural resources, and for improving the multi-functionality and sustainability of agriculture.

Examples of such agroecological and mainly 'know-how' based types of innovation include the use of composts in plant protection (Weltzien, 1991), encouraging predators by supporting their habitats (e.g. flowering field margins) (Wratten *et al.*, 2012), controlling parasites through knowledge of their lifecycle and thus reducing the building of infection (Cabaret *et al.*, 2014), finding natural sources of vitamins and minerals for livestock, such as herbs in pastures[20], or many of the other ecological examples described in this report. However, there are large gaps in our understanding of how to combine these essential components for the purpose of building effective, self-regulating systems.

It is, therefore, likely that the category '*understanding of the basic systems that sustain food production*' proposed by Garforth (2010) consists of scientific and technological knowledge as well as farmer (tacit) knowledge. In order to use the different types of knowledge effectively, the two main groups contributing to the agricultural knowledge system, farmers and scientists (and various intermediaries), need to work together. However, communication between both groups of actors is not easy, which may explain some of the difficulty in expressing clearly what farmers' knowledge is. This may be one reason why farmers sometimes have greater trust of other farmers and can be hostile towards outsiders that "*don't understand farming*". Koutsouris (2012) refers to this as the lay-expert gap.

Hill's (1985; 2014) 'Efficiency-Substitution-Redesign' (ESR) framework captures the progression from more shallow agroecological approaches to full system redesign. Hill argues that there are psychological as well as scientific and technological dimensions to addressing the challenges of large-scale farming, simplification and fragmentation. This leads to low resilience, change vulnerability and lack of sustainability, associated with 'denial of consequences' and problems understanding complexity over large space and timescales.

Caron *et al.* (2014) comment on the huge scale of the knowledge gap that needs to be overcome to address this. There is also a need for more research adopting a systemic approach to agricultural production, i.e. how the different components interact with each other and the impacts on human health, ethics and livelihoods (Garnett and Godfray, 2012).

With growing recognition of the role of farmers as active participants in agricultural knowledge systems comes a shift away from dissemination and 'technology transfer' towards learning, i.e. active knowledge construction (Koutsouris, 2012). For example, Moschitz *et al.* (2014) studied the role of Learning and Innovation Networks for Sustainable Agriculture (LINSAs). Such groups need to adopt a strong focus on the process of learning to effectively support innovation in the farming sector. In particular, the dimension of social learning with groups of farmers has received attention. In the LINSA groups, social learning emerged from a shared interest in a problem, challenge or activity, where all the actors contributed expertise. Social learning is linked to processes of trust building, trial and error and of mutual support. Social learning can be an answer to very complex problems, because mutual reflection on knowledge and consciously hearing different perspectives on one common issue will enhance the portfolio of potential solutions (Moschitz *et al.*, 2014).

Klerkx *et al.* (2012) refer to the importance, not only of farmers' learning groups, but also other players in the innovation process in agriculture and refer to a process of co-evolution between technological, social, economic and institutional changes. Caron *et al.* (2014) recognise the important contribution that science (and scientists) make to this process.

Knowledge production for agroecological intensification needs to find ways to effectively combine scientific and farmer knowledge and respect farmers as active and independent contributors, even if there is a tension between ensuring robustness while maintaining relevance to specific local conditions. There are many examples where farmers and scientists have worked together in different ways, such the Danish Stable Schools (Vaarst *et*

[20] ECOVIT see www.orgprints.org/view/projects/DA3-ECOVIT.html

al., 2007) for the reduction of antibiotic use in dairy farms, and the field labs of the Duchy Originals Future Farming programme (MacMillan and Benton, 2014).

3.5.2 Building closer links with consumers

In the context of agroecology, building closer links with consumers is considered important in several publications (e.g. Altieri, 2009; Marsden and Smith, 2005; Smaje, 2014). Practices involved include direct sales through farmers' markets, farm shops, box schemes and mail order, as well as new business models that directly involve the customers as partners (e.g. community supported agriculture). The creation of brands and/or production quality assurance schemes that are verified through certification is widespread. Links to local food action groups which may involve producers, consumers and civil society groups, which also function as learning groups for producers and consumers alike, are also common (e.g. Curry and Kirwan, 2014).

The most widely used formal approach for farmers and food producers to develop and provide special food attributes related to agroecological management is third party product certification. This is an important cornerstone of certified organic farming (with EU regulations and several private certification bodies operating in the UK), integrated agriculture (e.g. LEAF Marque) and many regional schemes. In the context of agroecology, process characteristics are often more important than product characteristics, including the farming and processing methods used, provenance (geographical indicators) and food authenticity. Most such additional attributes are defined by the private sector (farmers and food producers) and examples include several that refer to environmental outcomes, such as climate friendly food. The European Union has passed regulations in support of some voluntary schemes, mainly with the aim to protect consumers from false claims, such as certification of geographical origin (PGI/PDO, Regulation EC/510/2006), traditional specialities and production methods (TSG, Regulation EC/509/2006) and organic farming (Regulation EC/834/2007). Regulated certification schemes normally have public bodies for supervision, and any private third party certification bodies have to be accredited according to ISO 65/EN 45011.

There are many less formal approaches which, by their nature, are less well documented and their impact is harder to quantify. They may be instigated both by producers and by consumer groups. For example, Marsden and Smith (2005) studied two mainly producer-oriented groups related to meat marketing: one organic group in the Welsh Border country (Graig Producers); and one local branding initiative in the Netherlands (the Wadden Foundation) which created a local food label. They conclude that the personal initiative of key individuals is very important to realise benefits for producers through a market route. They use the term "ecological entrepreneurship" for the role of key actors in the networks that play a decisive role in enrolling and mobilising other actors into the network, create and sustain its structures, and innovate in developing new interfaces between producers and consumers. In Spain, where there is a very active engagement with agroecology as a movement, researchers identified more than 180 'horizontally-organised' food networks that aim at achieving a 'more sustainable' match between the production, distribution and consumption of food products. Of those more, than a 100 are mainly driven by consumers taking initiative to reshape their food access (Fernández *et al.*, 2012).

Smaje (2014) studied veg-box schemes in the UK to investigate the role of ethical consumption for promoting agroecology. He found that, for a small proportion of the population at least, consumption behaviour is significantly motivated by considerations of local seasonal food provenance, environmentally sustainable growing methods, a desire to have direct links with trusted small-scale farmers, and concerns over food safety in the conventional food system. However, there were strong countervailing tendencies drawing customers towards apparently cheaper produce and wider consumer choice, and pressurising producers to relax environmental standards.

Outside Europe, one example is the Californian wine industry which had used geographic branding linked to environmental and sustainability attributes to develop a specialist market. This carries the potential to enhance producers' incomes, but also to expose the specific circumstances of production to criticism on environmental grounds (Warner, 2007).

3.5.3 Social and cultural impacts

Cultural aspects of agroforestry systems, particularly in temperate regions, are often overlooked, despite the long tradition of systems such as woodland and orchard grazing, alpine wooded pastures, pannage, the dehesa and parklands (McAdam *et al.*, 2008). Lifestyles such as nomadism, transhumance (seasonal movement of people with their livestock), and traditional techniques such as pollarding and hedge-laying, are integrated within such systems and the symbolic and cultural perception of these landscapes are shaped by local practices, laws and customs (Ispikoudis and Sioliou, 2005). While only remnants of these traditional landscapes exist today, the significance and value of these cultural landscapes have been recognised at the international level by UNESCO[21] and at the European level by the European Landscape Convention[22]. Within the UK, national park status was awarded in 2005 to the New Forest, to protect one of the largest remaining areas of wood-pasture in temperate Europe.

Public attitudes to agroforestry reflect society's view of the non-market benefits connected with amenity, habitat, landscape and animal welfare. The visual impact of monocultures of crops in large scale arable fields or mono-species forest plantations is unappealing for many people; integrating trees into agricultural landscapes can increase the diversity and attractiveness of the landscape (McAdam *et al.*, 2008). However, as with all forms of tree plantings, modern agroforestry, characterised by rows of trees and alleys, may not be appropriate for all landscapes, particularly open landscapes such as downlands and fens, or historic landscapes such as parkland, moorlands and lowland heathlands. Agroforestry systems can provide recreational opportunities that can benefit the general public as well as the landowner (McAdam *et al.*, *op cit.*). Cultural landscapes such as the dehesas of Spain and Portugal, and the wood pastures of the Alps, can provide opportunities for eco-tourism.

3.6 Agroecological systems

Globally, hundreds of agricultural systems are based on agroecological principles – from rice paddies in China to some mechanised wheat systems in the USA, tropical subsistence agroforestry and perennial food-grain-producing systems.

The focus here is on systems relevant in a UK and European context and the primary purpose of this section is to highlight the origins and key characteristics of the systems, rather than to evaluate their performance, which is the subject of Chapter 4. Similar descriptions of the range of different approaches to agroecological systems can be found in Buckwell *et al.* (2014) and Scialabba *et al.* (2014).

3.6.1 Integrated pest management (IPM)

The concept of IPM came from research conducted by scientists at the University of California in the 1950s, which found higher levels of pest control in a crop of alfalfa where lower doses of insecticide were used (Stern *et al.*, 1959b). Closer investigation revealed that while the pesticide had effectively eliminated part of the pest population, beneficial species had survived and were able to impact upon the surviving pests. The concept of Integrated

---

[21] http://whc.unesco.org/en/culturallandscape
[22] http://www.coe.int/t/dg4/cultureheritage%20/heritage/landscape%20/default_EN.asp

Pest Management (IPM) was born, with the term first coined in 1967 by Ray Smith and Robert van den Bosch (1967).

Many of the ideas that make up IPM date back to some of the very earliest farming systems. What sets IPM apart is that it embraces both ecological and chemical pest control, attempting to integrate all forms of pest control, e.g. biological, chemical, cultural and mechanical control, as well as resistant varieties, in a long-term strategy which aims to minimise risks to the environment and human health. Critical to this is the management of the damage caused by pests using only the most economical appropriate controls. As such, a key facet of the IPM approach to pest management is the use of monitoring tools to better understand the pest infestation and to make informed decisions on the most sensible management response.

Successful IPM requires a detailed understanding of the population ecology of the pest, as only with this knowledge is it possible to make decisions about what pest control measures are appropriate and when to apply them. Understanding the population ecology of the pest and its relationship with environmental factors allows action thresholds to be set, which are the point at which control is required in order to minimise the potential economic damage caused. Because of the requirement for such a detailed understanding of the system, IPM strategies can be expensive to develop and require engaged management of the agro-ecosystem. In contrast to conventional management, careful monitoring ensures that pesticides are applied at the point when they are likely to be most effective (Emden and Peakall, 1996). This more judicious approach can be beneficial in reducing total pesticide usage and in doing so can help to reduce the development of pesticide resistance. IPM strategies are compatible with organic farming; in this case the management toolkit is altered to remove those elements that are not permitted under organic regulations.

The concept of IPM was adopted rapidly across the globe, often because it offered the only solution to resistant pest populations. In Northern European glasshouse production systems, it has reached a highly sophisticated level with the manipulation of the cropping environment though computer controlled climate management and the intentional and wholesale release of raised predatory and parasitic insects. While such an approach is perceived as being more "natural", and has significantly reduced pesticide use, it could in some cases be thought of as a far-cry from an agroecological approach, not least because many crops are grown in soil-less media, fed entirely on soluble inorganic fertilisers, with weeds controlled using polythene mulches, in a heated glasshouse atmosphere artificially enriched with carbon dioxide with controlled humidity.

3.6.2 Integrated crop/farm management (ICM/IFM)

Integrated Crop Management[23] (ICM) builds on the IPM concept, but takes a more holistic approach to crop management than IPM where the focus is more on individual pest species. There are many definitions for ICM, some with slightly different emphasis, but all of which are essentially "*to manage crop production on the whole farm in a way that maintains and enhances the environment for wildlife and people, while at the same time producing economic yields of high yielding, quality crops*" (ECPA, n.d.).

ICM was developed in response to some of the pest, weed and disease problems that had arisen from the intensification of crop production and the simplification of crop rotations, with

---

[23] In this section the terms Integrated Crop Management (ICM) and Integrated Farm Management (IFM) are used interchangeably, reflecting potential applications at both field and farm level. Integrated approaches to livestock management are, however, less well established and documented, therefore the ICM term arguably still reflects the current arable and horticultural focus of integrated management better and is used more frequently here.

increased nitrogen fertiliser use contributing to disease and other problems (Jordan and Leake, 2004). ICM is widely practiced in arable and horticultural cropping systems in the UK and elsewhere in Europe, to the extent that it is now argued in some cases to be "conventional farm practice", at least in terms of the minimum standards required of producers by multiple retailer buyers. The loss of key pesticide ingredients and the development of herbicide resistant weeds mean that an integrated management approach is now a necessity for successful cropping. There is also increased recognition of the importance of the ecological infrastructure on farms and wider aspects such as waste management and staff training. This has given rise to the concept of Integrated Farm Management (IFM), an approach championed in the UK by the charity Linking Environment and Farming (LEAF) who define IFM as: "*a whole farm business approach that delivers sustainable farming*" (LEAF). LEAF has also developed the LEAF Marque logo to support the marketing of IFM products in the UK. Other certification schemes including Conservation Grade and Good Natured, as well as the Red Tractor scheme, operate in the UK[24] and similar initiatives exist in other countries, but there is no European regulation defining ICM or IFM.

The development of ICM has been supported by research in the UK and elsewhere. However, researching systems approaches is problematic and costly, with replication difficult as soil types, crop rotations and seasons vary. Three Ministry of Agriculture (MAFF) trials focussed initially on pesticide use, responding to the development of IPM systems advocated by Stern *et al.* (1959a). The first, known as the *Boxworth Project*, ran from 1981-1991 and compared a full agrochemical spray programme with a more rational, so-called "supervised" approach. Importantly the project also examined the impacts of the pesticide regimes on predators, parasitoids and other invertebrates, as well as bird species and other environmental measurements (Greig-Smith *et al.*, 1992).

The Boxworth Project spawned two follow-on studies, carried out in the 1990's: *Seeking Confirmation about Results at Boxworth* (SCARAB) and *Towards a Lower Input System Minimising Agrochemicals and Nitrogen* (TALISMAN). This latter experiment included replicated small plot validation studies, as well as a whole field rotational approach, and crucially included a modified nitrogen fertiliser regime (Young *et al.*, 2001). In 1990, the British Ecological Society's 32nd Symposium "*The Ecology of Temperate Cereal Fields*" (Firbank *et al.*, 1992), paved the way for the development of agroecological approaches to pest management in arable rotations.

The *Less Intensive Farming and the Environment* (LIFE) project examined a 'conventional' arable rotation across six split fields over a period of 12 years and compared it with an 'integrated' system. Unique to this project was a study of the contribution that non-plough tillage can make to a range of environmental and production factors. The LINK *Integrated Farming System* project compared a conventional and integrated approach at six sites using different crop rotations on different soil types for a period of 5 years (Bailey *et al.*, 2003), and the *Focus on Farming* systems trials compared a conventional mixed farming system with rotational grass to an integrated and organic approach over a 12-year period (CWS, 2001).

The principle non-chemical control mechanisms exploited within an integrated approach are based upon cultural, biological, mechanical, physical and thermal manipulation. These are then supplemented by the use of chemical inputs. Ideally these inputs are rated by their environmental profiles so persistence, selectivity, leach-ability and volatility are all considered, along with the timing of the application, the prevailing weather conditions and the dose rate required to reduce the problem to below 'economically injurious levels'.

---

[24] For an overview in the context of CAP reform and Greening, see: http://www.scotland.gov.uk/Topics/farmingrural/Agriculture/CAP/regulations/Meetings-2013/Existing-Certification-Schemes

Integrated systems also pay closer attention to pest–prey densities than conventional systems, where often elimination of antagonists is seen as the objective. Economic thresholds are used to decide when a pest has reached levels requiring a controlled intervention. Stern *et al.* (1959a) defined the economic threshold as the "*density at which control measures should be determined to prevent an increasing pest population from reaching the economic injury level*". They defined the economic injury level as "*the lowest population level that will cause economic damage*" i.e. the cost/benefit ratio of the control measures rises above unity. Economic thresholds have now been evaluated for many crop pests to replace reliance on prophylactic treatments. This change of emphasis to applying pesticides only when necessary has led to typical reductions in pesticide use of 20-39% (Emden and Peakall, 1996).

The cost/benefit concept is based around those two components. The cost element is now recognised as being considerably more than the direct cost of the pesticides, labour and fuel involved in their application. The spray operation can cause crop damage through phytotoxicity, stem and leaf breakage, soil compaction and increasingly impacts on other natural regulatory process are considered as costs.

Weed thresholds are relatively easy to determine using glasshouse studies. However, straight weed density yield loss studies are complicated in the field by differing soil types, crop plant density and development, and interactions between different weed species. Furthermore, the legacy effect of permitting a low density population, which is below the economic injury level, to mature and shed seed can have serious consequences for their control in subsequent crops (Oerke *et al.,* 1994).

Crop rotation underpins the integrated approach. Typically in arable systems, broad-leaved crops are rotated with cereals. Different crops favour different species, with crops such as beans and oil seeds possessing tap rooted systems and producing insect-pollinated flowers, while cereals possess fibrous roots and are wind-pollinated. Plant sequences and sowing dates are manipulated to avoid the adverse effect of antagonists and to make best use of soil fertility. Delayed drilling in the autumn, for instance, allows the use of stale seedbeds which can either be cultivated or sprayed with herbicides. Where an early harvested crop such as vining peas, oilseed rape or winter barley is succeeded by late sown winter wheat, this extended inter-crop period allows a number of germination flushes and destruction events to occur, especially given adequate soil moisture.

The integrated approach has led the way in reduced cultivation systems. In 1999, less than 10% of the cropped land in the UK was subjected to non-inversion tillage, rising to 46% in 2005 (Basch *et al.*, 2008). However, despite this rapid growth, the percentage has stabilised at around 40% (Knight *et al.,* 2012). The development of non-persistent, contact-acting herbicides enables weeds to be managed in the inter-crop period and removes the need for deep ploughing. A wide range of benefits have been recorded, many of which are favourable to soil biota, wildlife generally, soil erosion and water quality in nearby watercourses. There are numerous reports of increased earthworm numbers (Hutcheon *et al.*, 2001), improved porosity (Jones *et al.*, 2006), decreased bulk density, and even increased microbial activity resulting in reduced erosion (Allton, 2006). Reduced cultivations are also implicated in increased numbers of beneficial predatory invertebrates (Jordan *et al.*, 2000a; Holland, 2004) and reduced pest attack (Kendall *et al.*, 1991). Increased porosity and the incorporation of crop residues at the surface have been shown to reduce run-off and soil erosion, improving water quality in nearby watercourses, and reducing transport of phosphate and nitrate (Jones *et al.*, *op cit.*).

Transects walked across a split-field comparison between plough-based cultivations and direct-sown crops showed birds strongly favoured the latter (Saunders *et al.*, 2000). Drilling depth in cereals is critical to avoid slug damage, with seeds sown at 20 mm showing 26% loss on emergence compared with just 9% at 40 mm (Glen *et al.*, 1990). When losses are limited to this level there is no need for slug pellets to be applied, which reduces exposure to non-target organisms, particularly where methiocarb is used.

Concerns that high levels of trash left on the soil surface result in the spread of diseases led to a review of existing literature, which concluded that the presence of inoculum is less of a contributory factor than the environmental conditions which give rise to infection (Leake, 2003). The perception that high levels of trash on the soil surface would lead to increased infestation of the subsequent crops is therefore unfounded.

3.6.3 Conservation agriculture (CA)

Conservation agriculture is a term used widely around the world, particularly in areas where soil is prone to erosion. It is widely practised in both North and South America as well as Europe. The system has three guiding principles:

1. The use of crop rotation
2. Minimum soil disturbance
3. Maximum soil cover

While the use of rotations has declined in conventional agriculture, there is renewed interest in response to concerns about soil health and weed control, in particular the control of blackgrass in the UK. In conservation agriculture, many crops are established using zero-till. This involves direct seeding into the stubble of the previous crop with no other soil disturbance taking place. Such a system is high speed, with lower cost and energy use, and retaining the soil in the field in a consolidated form helps prevent erosion.

Maximum soil cover is achieved by sowing immediately after harvest and chopping and leaving crop residues on the surface. Where there is an extended inter-crop period because the rotation dictates it, cover crops are grown. In some EU countries, notably Denmark, green cover crops are mandatory prior to the sowing of spring crops as part of their implementation of the EU Nitrate Directive. Research into the full scope of benefits brought from autumn sown cover crops, or catch crops as they are also known, is limited, and more research is required in order to determine the economic benefits compared to the cost of their establishment. Common cover crops in use and studied are mustard, fodder and oil radish, rye, and phacelia. Research to date suggests that their inclusion into sustainable integrated farm management rotations can prevent soil erosion, reduce nitrogen leaching by 65-70%, increase available nitrogen to the following main crop, increase soil organic matter, improve soil structure by remediating some compaction, reduce weed burdens and increase yields (Ebelhar *et al.*, 1984; Teasdale, 1993; Teasdale *et al.*, 1991; Kuo *et al.*, 1997; Hansen and Djurhuus, 1996; Wyland, 1996; Creamer, 1996; Teasdale, 1996; Snapp *et al.*, 2005; Dabney *et al.*, 2010).

Conservation agriculture in Europe is represented by the European Conservation Agricultural Federation (ECAF), which has members from 16 European states, with the UK represented by the UK Soil Management Initiative (SMI). Around 15% of crop land in Europe is in conservation agriculture.

3.6.4 Organic farming

Organic farming, whose origins (along with biodynamic agriculture) date back to the early 20th century, is commonly perceived as being primarily focused on the non-use of synthetic chemicals in agriculture. While this is (up to a point) a characteristic of the approach, it does not address what organic management involves instead - not using synthetic inputs and doing nothing else (organic farming by default) is likely to lead to failure in productivity, financial and environmental sustainability terms.

Organic farming can be defined as an approach to agriculture where the aim is to create integrated, humane, environmentally and economically sustainable production systems (Lampkin, 2003). This encompasses key objectives relating to achieving high levels of environmental protection, resource use sustainability, animal welfare, food security, safety and quality, social justice and financial viability. Maximum reliance is placed on locally or farm-derived, renewable resources (working as far as possible within closed cycles) and the management of self-regulating ecological and biological processes and interactions, in order to provide acceptable levels of crop, livestock and human nutrition, protection from pests and diseases, and an appropriate return to the human and other resources employed. Reliance on external inputs, whether chemical or organic, is reduced as far as possible in order to promote a self-reliant, self-sustaining system.

The term 'organic', first used in this context in the 1940s, refers less to the type of inputs used, and more to the concept of the farm as an organism (or system in modern terminology), in which all the component parts – the soil minerals, organic matter, micro-organisms, insects, plants, animals and humans – interact to create a coherent and stable whole. In many European countries, organic agriculture is known as biological or ecological agriculture, reflecting the emphasis on biology and ecosystem management. In other parts of the world, ecological or biological farming are sometimes used to refer to agroecological systems that are not certified organic.

The ideas and principles underpinning organic farming as a coherent concept go back almost 100 years (e.g. to King (1911); see also Lockeretz (2007)). Since then, different issues have come to the fore at different times, from soil conservation and the dustbowls in the 1930s (Howard, 1940; Balfour, 1943), to pesticides following the publication of *Silent Spring* (Carson, 1962), energy following the 1973 oil crisis (Lockeretz, 1977), and more recently to current concerns about animal welfare, biodiversity loss, climate change, resource depletion and food security. These ideas are reflected in the four fundamental principles of organic farming – health, ecology, fairness and care – defined by the International Federation of Organic Agriculture Movements (IFOAM, 2005).

The key elements of organic farming practice are described in Lampkin (2003); Lampkin *et al.* (2014) and in more detail in Lampkin (1990). Although the regulations focus on input restriction, particularly with respect to synthetic fertilisers, pesticides and genetically modified organisms (in part because inputs are easier to audit), organic farming practices include:

- Maintaining soil fertility using crops such as legumes and green manures;
- Conserving nutrients by aiming to close cycles, avoiding waste and unnecessary exports, and recycling nutrients where possible;
- Using relatively insoluble mineral nutrient sources, e.g. rock phosphate, in preference to high solubility or processed forms;
- Reducing energy use, and increasing reliance on renewable energy sources;
- Using shallow ploughing and reduced tillage techniques to protect the soil and its biological activity;
- Managing manures and slurries to conserve nutrients and avoid pollution, including through composting;
- Using crop rotations and polycultures to restore soil fertility, help control weed, pest and disease problems; provide sufficient livestock feed and maintain a profitable system;
- Replacing biocides for weed, pest and disease control with preventive cultural measures, supplemented by mechanical, thermal and biological controls if required;
- Integrating livestock with cropping systems (except in the case of stockless horticultural and arable farms), with both ruminants and non-ruminants ranging freely (i.e. no intensive, permanently-housed pig, poultry and feedlot cattle production);
- Relying as far as possible on home-grown feeds for livestock, limiting stocking rates to levels consistent with the EU nitrates directive, and thus reducing pollution risks;
- Mixing livestock species, such as sheep and cattle or sheep and poultry, to help control parasites and diseases and improve grassland management;

- Maintaining animal health through preventive management (including breeding, rearing, feeding and housing) and health plans in preference to prophylactic medication (e.g. with antibiotics or anthelmintics);
- Promoting animal welfare including freedom to exhibit normal behavioural patterns through housing design, stocking rate, restrictions on mutilations, but also through the use of conventional treatments if needed to avoid suffering from disease or injury.

The definition of organic farming, and the debate surrounding it, has been further influenced by the development of specialist markets for organic food since the 1970s, a relatively recent development in the history of organic farming (Lockeretz, 2007). In order to maintain the financial viability of organic systems, producers looked to consumers' willingness to pay higher prices for the perceived benefits of organic food. In some cases, the consumer interest reflected more altruistic environmental, animal welfare and social concerns; in others more 'self-interested' concerns relating to food quality and safety, in particular issues relating to pesticide residues and health. To protect consumers and *bona fide* producers, the development of the organic market involved the development of production standards, both at national level and globally.

As the market for organic produce developed, many countries, including the USA and the EU, introduced legal regulations, which required organic products to be certified before they can be marketed as such. The original EU regulation (EC, 1991) was substantially revised in 2007 (EC, 2007), in particular to include a clearer statement of the underlying principles of organic farming that might be used in future as a basis for determining acceptability, or otherwise, of specific practices. For many, these regulations have become the standard definition of organic farming, even though they contain some black and white distinctions, when in practice shades of grey may be more appropriate. The EU Regulations are being revised in 2015.

Building on the market potential and the environmental and other societal benefits attributed to organic farming, some EU Member States introduced specific policy support for organic farming from the late 1980s. This was extended on an EU-wide basis to support conversion to, and maintenance of, organic farming as an agri-environmental measure from 1994 (under EU Regulation 2078/92 and subsequent Rural Development regulations). The financial support provided recognises both the financial barriers to conversion and the costs of delivering environmental benefits using organic methods on a long-term basis, acknowledging that a minority of consumers paying a premium might not be sufficient to compensate fully for the income foregone.

The specialist markets and regulatory context of organic farming, particularly in the northern hemisphere, have led to some debate about whether some organic farms that achieve organic certification by input substitution rather than redesign (see section 2.2) can really be considered agroecological, and whether the organic concept has become a victim of corporate and political institutionalisation (e.g. Guthman, 2004; Lynggaard, 2006). Given the common heritage of the concepts, this debate may reflect more the different cultural and socio-economic contexts of the respective movements, with agroecology as a movement currently more associated with southern hemisphere contexts such as Latin America. Even so, there is a not insignificant risk that the market and the regulations can be seen as an endpoint for organic producers, rather than a step in developing sustainable systems based on organic/ agroecological principles.

3.6.5 Biodynamic agriculture

Biodynamic agriculture shares common roots with organic farming in the early part of the 20th century, with its origins in a series of lectures given by the Austrian philosopher and spiritualist Rudolf Steiner (1924). While many of the components of biodynamic farming, such as the restrictions on synthetic inputs, are similar to organic farming and subject to the same European regulations, greater emphasis is placed on the integration of livestock, in particular cattle and the use of composts rather than farmyard manures in the farming system. Biodynamic farmers view the ideal farm as a 'self-contained individuality' with the requirements for agricultural production coming from within the farm itself. The conceptualisation of the farm as an organism arguably goes further than the holistic, systems perspectives common to many other agroecological systems, as it is also influenced by Steiner's spiritual teachings. This idea of self-sufficiency is viewed as an ideal situation that cannot always be attained, but should be observed as far as possible. Koepf *et al.* (1976) also refer to a central concept of integration within biodynamics which "*extends to everything that belongs to the farm and lives in it - soils, livestock, crops, the people who work there, the wild plants, ponds and streams, wild birds and insects, wild animals, the local climate, the seasons and their rhythms, all of these elements interact as part of the whole*".

Biodynamic farming also differs from organic and other agroecological approaches in its use of preparations made from fermented manure, minerals and herbs, to help enhance the soil, improve the quality of compost and the quality, nutrition and flavour of the food produced (Biodynamic Association, 2014). These preparations were developed based on Steiner's lectures in 1924. They are not fertilisers themselves, but are intended to develop and assist natural processes and are used at very low rates per hectare. Biodynamic practitioners also recognise and strive to work in cooperation with the subtle influences of the wider cosmos on soil, plant and animal health (Sattler and Wistinghausen, 1992; The Biodynamic Association, 2014).

Many biodynamic farms seek to include triple bottom-line (ecological, social and economic) approaches to developing the farming system, taking inspiration from Steiner's thinking on social, cultural and economic life (Biodynamic Association, 2014). Community supported agriculture (CSA), for example, was originally developed by biodynamic farmers. As Moore (1997) highlights, "*...the CSA concept fits the biodynamic idea of a farm organism remarkably well. It closes that loop even tighter. When you bring the consumers in, it opens up all kinds of possibilities for recycling, compost programs -- all sorts of things.*" Many biodynamic farms are tied in with the Waldorf/Steiner School system of education. Some biodynamic farms also work in partnership with other farms and with medical and wellness facilities, restaurants, hotels, homes for social therapy and other organisations (Biodynamic Association, 2014).

3.6.6 Agroforestry

"Agroforestry is the practice of deliberately integrating woody vegetation (trees or shrubs) with crop and/or animal systems to benefit from the resulting ecological and economic interactions."[25]

Agroforestry combines elements of agriculture (agro) and trees (forestry) in sustainable production systems combining the production of a wide range of products including food, fuel, fodder and forage, fibre, timber, gums and resins, medicinal products, recreation and ecological services (Dupraz and Liagre, 2008). Agroforestry has been identified as a 'win-win' multi-functional land use approach that balances the production of commodities with non-commodity outputs such as environmental protection and cultural and landscape amenities (McIntyre *et al.*, 2009). Key characteristics that distinguish agroforestry systems from agriculture and plantation forests include greater structural and functional complexity, an emphasis on multipurpose trees, and the production of multiple outputs balanced with protection of the resource base (Nair, 1991).

Agroforestry systems have traditionally been important elements of temperate regions around the world, evolving from systems of shifting cultivation towards more settled systems

[25] www.agforward.eu

involving agriculture, woodland grazing and silvopasture, with fertility transfer from woodlands to cultivated land via manure (Maydell, 1995; Eichhorn *et al.*, 2006). European examples include:

i) Agroforestry systems of high nature and cultural value such as Dehesas in Spain, and wood pastures and parklands across Europe. Such systems also include hedgerow systems in northwest France and the UK;
ii) The intercropping and grazing of high value tree systems such as grazed orchards or the intercropping of olive orchards;
iii) agroforestry for arable systems such as alley cropping systems;
iv) agroforestry for livestock systems such as woodland eggs and poultry or the combination of trees with pigs, cattle, or sheep.

Traditional agroforestry systems in the UK include: wood-pasture, such as the New Forest, which features some of the oldest and widest trees in Europe, providing valuable resources for a wide range of associated biodiversity, as well as having historical and cultural value (Isted, 2005); parklands, which were developed in 18th century Britain for aesthetic reasons, but the economic value of their open grown timber for ship building was subsequently recognised (Sheldrick and Auclair, 2000); and multi-functional hedgerows.

Modern commercial agroforestry systems such as silvoarable and silvopastoral systems integrating trees into agricultural systems are in their infancy in the UK, but becoming increasingly widespread across a number of European countries where policy support is available for establishing new systems (e.g. an estimated 3000 ha per year established in France). However, agroforestry research sites have a longer history within the UK, with networks of silvopastoral and silvoarable trial sites established in the late 1980s/early 1990s. Originally established to investigate agroforestry as a means of taking land out of production or reducing productivity during a period of food surpluses (Hoppe *et al.*, 1996), it was soon apparent that agricultural production could be maintained despite the introduction of trees so funding for agroforestry research was gradually reduced (Sibbald, 2006).

Perhaps the most commercially successful example of agroforestry in the UK is the production of '*Woodland Eggs*'[26]. Organic and non-organic free range eggs and chickens are produced from approved 'woodland farms' where chickens have access to woodland. Through a partnership between Sainsbury's and the Woodland Trust, there are now almost 200 farms involved and, since 2004, over 300,000 trees have been planted. Farmers must comply with strict guidelines to receive a bonus payment of 2p for every dozen eggs; the Woodland Trust also receives a donation of 1p per dozen eggs and 2p per chicken sold. Recognising the animal welfare benefits of increasing tree cover, in 2007 and early 2008, all laying hen producers within the McDonald's Restaurants UK supply base (286 producers) were required to plant, if not present already, 5% of the total range area in trees (Bright *et al.*, 2011; Bright and Joret, 2012). The Pontbren Project, a farmer-led initiative that used woodland management and tree planting to improve the efficiency of upland livestock farming within one of the wettest areas of the UK, has been a highly successful example of the multiple benefits of integrating trees and woods into farm management (Woodland Trust, 2013). A group of ten farmers managing a total of 1000 ha within the Pontbren catchment near Welshpool came together in 2001 to make their businesses more sustainable by planting more than 10 miles of hedges and 120,000 trees and shrubs to provide shelter for livestock. It soon became apparent that tree planting not only benefitted the farm business and wildlife habitats but also reduced water run-off during heavy rain, and the project became the focus of scientific research into the effects of land use in catchments prone to flooding (Jackson *et al.*, 2008).

---

[26] http://www.woodlandeggs.co.uk/

There are a number of unique agroforestry systems that have been established by UK farmers over the last 15-20 years, each system designed to meet specific needs and challenges.

Wakelyns Agroforestry, run by Martin and Ann Wolfe, is a diverse organic agroforestry system in eastern England which incorporates four silvoarable systems; short rotation coppiced (SRC) willow, SRC hazel, mixed top fruit and nut trees, and mixed hardwood trees with 10-12 m-wide crop alleys between tree rows. The reasons behind establishing such a diverse system were manifold: to reduce pest and disease pressure by increasing the distance between individuals of the same species; to increase biodiversity including beneficials such as pollinators and natural enemies; to provide resilience to a changing climate; and to diversify production and reduce the risks associated with farming single commodities.


Arable agroforestry at Wakelyns, Suffolk (Photos: ORC (L), Permaculture Association (R))

Whitehall Farm is a 120 ha arable farm on Grade 1 fen soil near Peterborough, Cambs, managed by tenant farmers Stephen and Lynn Briggs. Previously managed as an intensive arable system, the eastern half of the farm entered into organic conversion in 2008, while the rest entered conversion in August 2009. The western 60 ha was developed into an apple orchard/crops agroforestry system in October 2009 with 4,500 apple trees, consisting of 16 varieties (10 commercial and 6 traditional) planted in rows (NE/SW orientation) 27m apart, with 3m spacing of trees within rows. The understorey was sown before tree planting with a 3m band of nectar flower mixtures, wild bird seed mixtures and legumes. The 24 m remaining between rows is cropped on an organic rotation that includes cereals, field vegetables and fertility-building leys. The drivers behind planting agroforestry include Stephen and Lynn's desire to increase the enterprise diversity away from just cereals, to reduce soil erosion and to increase biodiversity.


Apple and cereal agroforestry at Whitehall Farm, Peterborough (Photos: J Smith/ORC)

Shillingford Organics is situated on a 140 ha farm just three miles from the centre of Exeter, owned and managed by Martyn Bragg. 18 ha are dedicated to growing organic vegetables, mostly for an organic vegetable box scheme where subscribers buy a box of mixed vegetables, herbs, salad leaves, edible flowers and some fruit once per week. Recently tree strips with apple trees have been introduced into the vegetable plots with the aims of increasing biodiversity, sheltering early vegetable crops, increasing the variety of crops to sell, and to provide interest for customers visiting the farm.

Grain legumes, crimson clover with trees at Shillingford Organics (Photos: J Smith/ORC)

Martin Crawford started establishing a forest garden on a 0.85 ha field at **Dartington Estate** in South Devon in 1994. The forest garden is now an excellent demonstration of a self-sustaining system with a highly diverse range of species which produce fruits, nuts, fungi, edible leaves, medicinal products, timber and other woody products. The system is robust to pests and diseases and resilient to the climatic extremes experienced over the last few years. Martin has written widely on his experiences, in particular Crawford (2010).

Martin Crawford demonstrates his forest garden at Dartington (Photo: J Smith/ORC)

Andrew and Hilary Mylius run **Brackmont Farm and Forestry** at the St Fort Estate in Fife. This mixed arable/grassland/woodland estate of 600 ha has pedigree sucklers, one herd of 85 pedigree Aberdeen Angus and one herd of pedigree Lincoln Reds. Trees were first planted on the estate in 1972 with the aim of providing shelter for the cattle and the farmhouse, and to reduce soil erosion. Further tree planting followed in 1994 to provide more shelter as amenity plantations for landscape and shooting. These woodland patches have areas of highly productive pasture, and so the cows and calves are moved in and out as the weather dictates, to take advantage of the grass growth and shelter from the trees. The ambience and sporting use of the estate has improved, and the suckler herd has been expanded without new sheds being required. The Scots pines are expected to yield saw logs in the future.

Livestock sheltering under pasture agroforestry, Brackmont Farm (Photo: J Smith/ORC)

There is increasing evidence on the potential benefits of agroforestry in temperate developed countries (Jose, 2009; Smith *et al.*, 2012; 2013a; see also Chapter 4). In particular, such systems have the potential to increase total biomass production, as measured by Land Equivalent Ratio (LER; see section 4.2), whereby the yield from growing crops in combination is greater than if the two crops were grown separately. Trees typically have longer growing seasons than annual crops and can capture more solar energy and nutrients during the year as a result.

The impact of agroforestry on the environment occurs at a range of spatial and temporal scales; externalities from farming systems impact the environment and society at regional or national scales. Agroforestry systems are multifunctional but most research focuses on a single function. One of the few studies to consolidate the multiple services from a single agroforestry system reports on ten years of research on agroforestry strips of hybrid plane (*Platanus hybrida*) and the shrub *Viburnum opulus* in north-east Italy (Borin *et al.*, 2009). The young tree strips reduced total runoff by 33%, N losses by 44% and P losses by 50% compared to non-buffer controls, while a mature buffer reduced both NO_3-N and dissolved phosphorus by almost 100%. Herbicide abatement was between 60 and 90% depending on the chemical and time since application, and it was calculated that the buffer strips sequestered up to 80 t C ha^{-1} yr^{-1}. The tree strips caused negligible disturbance to maize, soybean and sugarbeet yields, and contributed to increasing the aesthetic value of the landscape based on a visual aesthetic index formulated from people's preferences during interviews (Borin *et al.*, 2009).

Field experiments have by necessity been restricted to studying the point-scale or farm-scale effects of incorporating trees and crops/livestock; modelling approaches have provided a means of extrapolating how these fine-scale processes impact on a landscape-, regional- and global-scale. Palma *et al.* (2007b) showed through computer simulations that silvoarable agroforestry in Europe could reduce soil erosion by up to 65%, reduce nitrogen leaching by up to 28%, sequester between 5 to 179 t C ha^{-1} and increase landscape biodiversity by an average factor of 2.6.

Porter *et al.* (2009) calculated the values of market and non-market ecosystem services of a novel combined food and energy agroforestry system in Taastrup, Denmark. Belts of fast-growing trees (hazel, willow and alder) for bioenergy production are planted perpendicular to fields of cereal and pasture crops, and the system is managed organically with no inputs of pesticides or inorganic N. Field-based estimates of ecosystem services including pest control, nitrogen regulation, soil formation, food and forage production, biomass production, soil carbon accumulation, hydrological flow into ground water reserves, landscape aesthetics and pollination by wild pollinators produced a total value of US$1074 ha^{-1} of which 46% is from market ecosystem services (production of food, forage and biomass crops) and the rest from non-market ecosystem services. Porter *et al.* (2009) then extrapolated these values to the European scale and calculated that the value of non-market ecosystem services from the system could exceed the value of European farm subsidy payments in 2009.

Despite these benefits, both farmers and policy-makers have reservations about the potential for agroforestry systems. As part of the EU-funded AGFORWARD project[22], 40 stakeholder groups[27] have been established across Europe, and farmers were asked to identify the key positive and negative issues associated with the system. While they supported many of the benefits identified here, they also highlighted concerns about complexity (and the impact of this on management), labour and machinery costs.

[27] http://www.agforward.eu/index.php/en/FarmerNetworks.html

3.6.7 Permaculture

Permaculture ('permanent agriculture') has its roots in agroforestry, but has become much broader in its conceptualisation, building on the original writings of Mollison (1990; Mollison and Slay, 1994) and Holmgren (2011). A key feature of permaculture is the emphasis on 'Permaculture design', which looks not only at the integration of plant and animal species to maximise the potential for beneficial interactions between them, but also at the design of the holding (including the household) to maximise energy and labour efficiency. The underlying principles are intended to be applicable in any climate and at any scale. Commercial permaculture operations have been developed in Australia, New Zealand and North America, and urban agriculture versions have been promoted in some developing countries (e.g. Kenya). Permaculture applications in the UK have tended to focus on smaller, non-commercial holdings[28].

The UK Permaculture Association[29] summarises the core principles of permaculture (based on Holmgren (2011)) as:

1. Observe and interact
2. Catch and store energy
3. Obtain a yield
4. Apply self-regulation and accept feedback
5. Use and value renewable resources and services
6. Produce no waste
7. Design from patterns to details
8. Integrate rather than segregate
9. Use small and slow solutions
10. Use and value diversity
11. Use edges and value the marginal
12. Creatively use and respond to change

While at first sight these may appear more distanced from the agroecological concepts covered in this report, in practice they reflect an emphasis on applying ecological design principles to food production and sustainable living, many of which are also reflected in organic farming and other approaches. These include the emphasis on: using biological resources; cycling of energy, nutrients and resources; accelerating succession and evolution in cropping system design; and the emphasis on diversity and complexity of relationships in and between system components. As in other approaches, information/knowledge intensity is also emphasised, but with perhaps a stronger focus on creative imagination and design.

3.6.8 Other systems

Several other variants of agroecological systems are prevalent in the literature, including: low (external) input sustainable agriculture (LISA/LEISA); regenerative agriculture; renewable agriculture; ecological agriculture and biological agriculture. These all share common perspectives, but also reflect the specific context in which the terms have been popularised.

In the USA, LISA was promoted as the integrated farming alternative to organic production for many years, but LEISA was popularised more in the context of developing countries with a focus on resource-poor farmers.

Regenerative agriculture was coined in the USA by Robert Rodale (whose father popularised organic farming and gardening and set up the Rodale Institute) as an attempt to

---

[28] For example, http://landmatters.org.uk/; http://www.permaculture.co.uk/articles/need-large-scale-permaculture-farms

[29] https://www.permaculture.org.uk/

overcome negative attitudes towards organic farming at the time. The term renewable agriculture picks up similar themes, emphasising the capacity for most resources used by agriculture, and in particular soils, to be renewed or regenerated if managed appropriately.

Biological husbandry, biological agriculture, eco-farming and ecological agriculture are also variations on a theme – in some parts of the world, particularly North America, the terms may be used as a point of differentiation from organic, because of dissatisfaction with the constraints and limited scope of organic regulations. In Europe, however, the terms biological and ecological (and the abbreviations eco and bio) are used and regulated as synonyms for organic farming.

At heart, all the approaches described in this section share significant common ground in terms of the issues that they are attempting to address, and the agroecological principles applied to do this.

4 EVALUATION OF AGROECOLOGICAL APPROACHES TO SUSTAINABLE INTENSIFICATION

4.1 Introduction

In this Chapter, we draw on a combination of grey and peer-reviewed literature, other web-based resources and quantitative data where available, to describe and assess the performance of agroecological systems and strategies compared with more conventional approaches to sustainable intensification.

Any assessment of performance requires the identification of relevant objectives, related outputs or indicators of performance, and criteria against which success or failure of different systems can be determined. In this context, there are a very wide range of possible objectives, systems, metrics and indicators with variable data quality and comparability, so inevitably some constraint to the assessment, and reliance on judgement, is required.

Elliott *et al.* (2013), Garnett *et al.* (2013) and Buckwell *et al.* (2014) provide some sources of relevant indicators. In addition, the Defra-funded, Warwick HRI-led project on developing methods for Farm System Sustainability Assessment generated a range of environmental, social and economic indicators that are highly relevant to this discussion (Lillywhite *et al.*, in progress). These are summarised in Table 4-1.

Several of the report reviewers and experts at the workshop emphasised the importance of using the correct metrics and quality data in such assessments. The work that is ongoing as part of Defra's Sustainable Intensification Platform (Defra, 2014a) and elsewhere will be critical for future work. The issue of the interchangeability of numerators and denominators was also raised (e.g. food produced per unit of GHG emissions or GHG emissions per unit of food produced), where the choice might be influenced by policy priorities. The relevance and importance of the denominator may also be site or time dependent, for example extracted water use per ha or per kg food produced might be critical in areas where availability is limited but not in high rainfall regions. Equally, if resources other than land are limiting, then yield per unit of the limiting resource may be more important than yield per ha.

Given the potential complexity of the evaluation, we have restricted the scope to five of the primary objectives identified in Table 4-1:

i. Productivity
ii. Carbon sequestration, greenhouse gas emissions and energy use
iii. Biodiversity and related ecosystem services
iv. Soil and water resources (physical aspects)
v. Profitability

The specific metrics prioritised are identified in each of the relevant sections below.

We have also focused our analysis on three specific groups of agroecological approaches: integrated crop management, organic farming and agroforestry.

At the end of each of the five main sections, a table summarises the key factors influencing the performance of each of these three approaches with respect to key sustainability/sustainable intensification outcomes. Given the nature of this study, and the wide range of unrelated studies drawn upon, the scorings are based on the authors' judgement rather than a prescribed methodology.

In the concluding discussion we attempt, as far as possible, to examine interactions between the different primary objectives and assess how some of the major agroecology practices, such as polycultures or use of biological nitrogen fixation using legumes, can contribute to sustainable intensification.

Table 4-1: Indicators of performance relevant to sustainable intensification from previous studies

Primary objectives	***Elliott et al. (2013)***	***Garnett et al. (2012; 2013) (mentioned in discussions)***	***Buckwell et al. (2014) (review of sustainability metrics)***	***Warwick HRI (Lillywhite et al., in progress)***
Productivity/ Intensity Land Equivalent Ratio; Total Factor Productivity	Output (food gross energy)/ha including uncropped land and land used to produce feed bought in	Productivity (production/unit of key inputs) rather than production *per se* (environmental factor productivity or eco-efficiency)	Population/ha Tractors/ha Gross capital stock/ha Fertiliser and pesticide use/ha	Land use Net System Output[27] Output diversity Business autonomy/ resilience
Profitability/ economic factors	(Value of food sold/ha) – not explicitly addressed	Not covered in detail	Production value/ha Subsidy dependence	Farm business income Net worth Net investment Return on capital Resource use/ £ output Land and capital costs
GHG emissions/ energy use/ climate/ air	Pollutants (modelled): ammonia; Carbon footprint (CALM) – CO_2, NO_x, CH_4	GHG emissions Non-renewable resource use	Emissions Energy use/ha	Fossil energy use Indirect energy use Ammonia emissions GHG emissions
Biodiversity/ Landscape/ Ecosystem services	Not directly measured Habitats, rare breeds and agri-environm. scheme participation as proxies, also crop diversity, non-cropped habitats	Established biodiversity indicators, pollinators	Farmland birds Pesticide impacts Landscape elements	Biodiversity/ landscape value
Soil/Water	(Modelled) Nitrates Phosphates Sediment Pesticides	Water quality	Soil erosion Soil organic matter/carbon Nitrogen balance Nutrient/ pesticide run-off/ leaching Water use/ha	Abiotic resource use Water use Pesticide use Acidification and eutrophication potential
Animal welfare/ ethics	Not covered	Yes	Not covered	Animal welfare
Social factors	Not covered	Not covered	Age Education Hired labour use Knowledge/ha	Labour use /ha Demographics composition Social capital Governance Succession
Human health/ nutrition	Not covered	Not covered	Not covered	Not covered

4.2 Productivity

4.2.1 Assessing system productivity

As identified in Chapter 2, productivity is a key element of sustainable intensification. Can agroecological approaches, which may offer significant environmental benefits, also deliver against the productivity benchmark and meet food security goals? Or would it be better to focus intensive production and environmental conservation on separate areas of land, as argued by some in the land sharing versus land sparing debate (see Adams, 2012; Tscharntke *et al.*, 2012; Balmford *et al.*, 2012 in Herzog and Schuepp, 2013; see also Section 4.4.5 below)?

At its simplest, the assessment of productivity can involve a focus on the total production or yields of individual crops and/or livestock per unit of land area, which assumes that land is the most limiting resource against which output should be measured (if land is not limiting, for example in arid environments, then other variables such as water, energy or labour may be relevant denominators). However, productivity also implies efficiency with respect to resources used (and their related emissions), which may involve a trade-off between yields per hectare and, for example, fossil energy use and GHG emissions per kg of food produced. The broader the scope of the sustainable intensification concept, the more the risk of trade-offs between individual components and objectives. Total Factor Productivity (TFP), which takes account of all resources used, not only land, labour and capital, is one approach to addressing this, although we have not been able to identify relevant studies that measure this on a comparative basis between different management systems, although Melfou *et al.* (2007) do consider some of the methodological issues.

Measuring system productivity is further complicated by considerations of joint production, whether this is crops grown in combination, as in agroforestry, or the production of non-commodity outputs, including ecosystem services other than provisioning ones. In addition, some outputs, e.g. livestock feeds, are used as inputs to other production enterprises, and therefore net output rather than total output may be more relevant in terms of meeting human needs (see also Cassidy *et al.*, 2013). A number of solutions have been proposed to address these issues.

- Land Equivalent Ratios can be used to assess the productivity of polycultures versus monocultures (Mead and Willey, 1980). The LER method compares the total yield of the polyculture components with the yields of the components if grown separately. A value greater than one indicates a net benefit from the polyculture. This measure is used in the assessment of agroforestry below;
- Net System Output (NSO) is a concept developed by ORC as part of the Warwick-HRI study (Lillywhite *et al.*, in progress) to permit the assessment of the productivity of whole farming systems (as opposed to individual commodities), enabling system-level resource use efficiency and emissions to be assessed. It is a measure of total production of commodities from a system, net of outputs used as inputs to other activities, e.g. cereals fed to livestock[30]. This may provide a better means of

[30] Net system output can be calculated using energy (GJ) and nitrogen (kg) outputs for commodities produced, adjusted for crops and crop by-products used as inputs, both home-produced and imported. The key variables of energy and nitrogen represent human nutritional and fuel requirements (including protein), and link to carbon and nitrogen cycles relevant to climate change and resource use sustainability. Using standard energy and nitrogen values for wheat, it is possible to calculate a ‘wheat equivalent’ value (tWeq) analogous to tCO_2eq values so that all commodities produced can be assessed on the same basis. This method does not currently permit the inclusion of other environmental goods and ecosystem services as relevant outputs.

assessing ability of systems to meet defined human needs per ha of land or other unit of input (see also Cassidy *et al.*, 2013)[31];
- Gross energy output was used by Elliott *et al.* (2013) and is similar conceptually to the NSO concept;
- Financial output measures can also be a good means of contextualising productivity, particularly if comparisons are being made between very different land uses, such as upland livestock and protected cropping.

We also consider the issue of labour productivity and farm scale from different perspectives, as these social issues are an important part of the agroecology debate, where some commentators consider high employment potential and small farm size an advantage rather than a disadvantage.

4.2.2 Integrated crop/farm management

The availability of data comparing integrated and intensive conventional systems is limited, both due to the lack, in the early stages of its development, of specifically-identified integrated farms to survey, and more recently due to the increasing convergence of integrated and other conventional systems.

Few studies have focussed directly on yield variations between cultivation techniques alone, as most of the system studies have tended to adopt an integrated approach which includes other agronomic changes (Ogilvy, 2000). However, where studies have been done, there is an indication that yields are reduced by between 1 and 7% in alternative tillage systems compared to ploughing. However, this has tended to be in 'like for like' comparisons and does not take into account the fact that a greater proportion of an autumn-sown crop is likely to be sown early in good conditions as a result of faster field operations. Trials at the Focus on Farming Project, which included a seven split field comparison, showed that wheat drilled in mid-September yielded around 1.0 t/ha more than that sown in late October (Leake, 1995).

In Europe, two experimental farms were established using the IOBC[32] Integrated Production principles. In the Netherlands, a farming system comparison was established at Nagele comparing organic, conventional and integrated approaches. Gross margins for the integrated system were reported at between 5 and 8% higher than the conventional system (Wijnands, 1997). At Lautenbach, in south West Germany a 'supervised' approach was compared with conventional farming. Between 1979 and 1981, yields were 98% of the conventional system, but by 1995/96 were 15% higher (Pretty, 1998).

The Swiss long-term DOK (bioDynamic, Organic, Konventional) comparison trial dating from the 1970s (Mäder *et al.*, 2006) has compared the performance of conventional, integrated, organic and biodynamic systems, albeit on the basis of similar rotations. The conventional (mineral fertiliser only) and integrated (mineral and organic fertiliser) high input treatments produced comparable yields.

Although Elliott *et al.* (2013) quantified gross output (measured as GJ energy produced) on a range of different case study farms representing different farm types, including some under integrated and organic management, it is difficult to draw conclusions relating to the relative performance of integrated farms.

---

[31] One reviewer argued that demand-side factors such as human food needs should not be included in such metrics, as demand can be affected by a wide range of economic and other factors. However, the basic minimum requirements for human existence, in terms of energy, protein and other key nutrients, have been defined by FAO and others, and are considered central to the food security debate. Cassidy *et al.* (*op cit.*) focus also on these minimum needs.

[32] http://www.iobc-wprs.org/index.html

With respect to labour productivity, all of the 12 commercial farm case-studies featured in the Soil Management Initiative's *Crop Establishment Guide* (SMI, 2005) showed substantial reductions in work days under reduced tillage, often over 50%. Where these studies were combined with local rainfall data, very often there were insufficient work days available for all crops to be established in good time where ploughing was used as the principle cultivation method, and this would have had a marked effect on yield.

To the extent that there is any yield reduction from integrated management, it is argued that the costs are offset by the increased efficiency of input use, including both agrichemicals and labour, but there may well be opportunities for increasing both yields and efficiency using this approach.

4.2.3 Organic farming

Low yields are perceived to be the key disadvantage of the organic approach, although the reductions compared with conventional systems reported in different studies have been highly variable. In the UK, organic wheat yields are typically little more than half those of conventional systems (Table 4-2; Moakes *et al.*, 2013; 2014; Lampkin *et al.*, 2014). However, this reduced productivity is exacerbated due to the need for fertility building crops in the rotation, so that organic farmers cannot grow wheat every year. Therefore the additional land area required to grow a tonne of wheat may be higher than a simple comparison of relative yields would suggest.

Table 4-2: Relative organic and non-organic (conventional) yields from Farm Business Survey data for England and Wales, 2011/12

Product	***Unit***	***Organic (n farms)***	***Non-organic (n farms)***	***Relative %****
Winter wheat	t/ha	4.4 (37)	8.3 (272)	53
Spring barley	t/ha	3.8 (44)	5.3 (136)	72
Winter oats	t/ha	4.1 (17)	6.4 (37)	64
Field beans	t/ha	2.8 (26)	3.9 (59)	72
Potatoes	t/ha	29 (6)	44 (23)	66
Milk	l/cow	6315 (45)	7397 (145)	85
Stocking	LU/ha	1.4	1.7	82
Milk	l/ha	8841	12575	70

*Organic as a percentage of non-organic
Source: Moakes *et al.* (2013)

Similar comparative data are not currently available for Scotland. However, research by Watson *et al.* (2011) and Taylor *et al.* (2006) has evaluated organic yields. Watson *et al.* found that annual grain yields of organically grown oats following a ley in a system trial near Aberdeen were not significantly different from National List yields in northeast Scotland for oats receiving 100 kg N ha^{-1} but no fungicides. Taylor *et al.* (*op cit.*) reporting on the same systems trial, found organic oat yields to be higher following the ley than when oats were grown later in the rotation, and that while there were annual fluctuations in cereal yields, there was no evidence for declining yields or soil organic carbon levels over time.

Three recent meta-analysis studies have reviewed the global evidence on organic crop yields. Ponti *et al.* (2012) analysed data from 362 studies concluding that organic crop yields are on average 80% of conventional yields, but finding significant regional and crop type variations, with organic yields ranging from 20% to 177% of conventional. Seufert *et al.*

(2012) found average organic crop yields to be 75% of conventional, with only 5% differences for rainfed legumes and perennials. Both studies make reference to an earlier much debated review by Badgley *et al.* (2007) who concluded that organic yields were 30% higher than conventional in a developing country context.

Ponti *et al.* (2012) identified, as other studies have done, that the organic–conventional yield gap increases as conventional yields increase, but this relationship was not strong. They hypothesised that when conventional yields are high and relatively close to the potential or water-limited level, nutrient stress must, as per definition of the potential or water-limited yield levels, be low and pests and diseases well controlled, which are conditions more difficult to attain in organic agriculture. Seufert *et al.* (2012) suggested that with good management practices, particular crop types and growing conditions, organic systems can nearly match conventional yields. It is clear from all these studies that yield differences found for specific crops in specific regions cannot be generalised globally.

The most recent meta-analysis (Ponisio *et al.*, 2014), using data from 115 studies, found that organic yields were, on average across all crops, 19% lower than conventional. While most individual crops types showed a similar organic yield reduction to the average, perennial fruit and nuts yielded closer to conventional, while root crops showed a bigger yield gap. They also found that multi-cropping (polycultures) and crop rotations when applied only in organic systems could substantially reduce the yield gap.

These results are consistent with the DOK long-term comparison between conventional/ integrated, organic and biodynamic farming in Switzerland (Mäder *et al.*, 2006), which found organic yields on average 20% lower than conventional, ranging from up to 42% reduction for potatoes, 33% reduction for wheat to 11% for forage crops and parity for soybeans.

Using the Net System Output approach (see Section 4.2.1 above) applied to data for different farm types from the English Farm Business Survey (FBS) in 2009, Lampkin *et al.* (in Lillywhite *et al.*, in progress) found that organic specialist cereal, general cropping and mixed farms (Table 4-3) performed less well relative to non-organic than did dairy farms (Table 4.4) in terms of tonnes wheat equivalent (tW_e) produced per ha. This may be related to Elliott *et al.*'s (2013) finding that arable farms generally had much higher gross energy outputs per ha than dairy farms. The values include the impact of using some land for fertility building in organic farming, and therefore reflect the challenge of utilising the fertility-building phase of the rotation effectively on cropping-oriented farms. However, performance with respect to farm business income per tW_e produced and tW_e produced per £ spent on inputs were both higher than conventional, despite the lower output per hectare. Greenhouse gas emissions were similar between organic and conventional with respect to tW_e produced.

It is worth reflecting on the extent to which yield differences can be explained by the relative nitrogen dependency of conventional systems. This would explain why wheat yields in the UK, where conventional N inputs are high, show larger differences than in some of the other studies. Figure 4-1 shows how UK wheat yields (non-organic) have varied with nitrogen use since the mid-1970s, with the lack of recent further yield growth associated with a flatlining of nitrogen fertiliser use at ca. 200 kg N ha^{-1} since 1980 (Sylvester-Bradley *et al.*, 2008). UK organic wheat yields, at 4-5 t ha^{-1}, are similar to conventional yields in the mid 1970s with nitrogen use levels ca. 100 kg N ha^{-1}, and much higher than pre-war yield levels when no fertilisers were used. In the US, where conventional wheat is produced less intensively with average yields about 3 t ha^{-1}, studies show more similar yields. Within the UK, yield differences for crops such as oats and field beans, where less N is used conventionally, are also lower. Legumes in general, in part because of the improved utilisation of biologically-fixed nitrogen, show organic yields closer to conventional.

Table 4-3: Net system output and other performance measures for organic and conventional cereal, cropping and mixed farm types, 2009.

Description	*Cereals (C)*	*Cereals (O)*	*Cropping (C)*	*Cropping (O)*	*Mixed (C)*	*Mixed (O)*
Farms (n)	356	20	199	18	127	16
Size (ha)	248	168	225	198	158	225
FBI (£/ha)	225	242	314	386	245	298
Input (£/ha)	341	66	420	170	202	76
NSO (tW_e/ha)	6.9	3.0	11.1	5.0	7.6	3.7
FBI (£/tW_e)	33	81	21	78	32	80
Efficiency (kgW_e/£In)	20.2	45.5	26.5	29.3	37.8	48.8
GHG emissions (kgW_e/ $kgCO_{2e}$)	3.4	3.4	3.3	3.5	1.4	1.5

Key: C=conventional; O=organic; FBI=farm business income; Input(In)=fossil-fuel based crop inputs (ferts, sprays, diesel) and water; NSO=net system output; We=wheat equivalent (excluding residual N); GHG=greenhouse gases; CO_{2e}: Carbon dioxide equivalent including methane, nitrous oxides.
Source: Lillywhite *et al.* (in progress)

Table 4-4: Net system output and other performance measures for high, medium and low intensity organic and conventional dairy farm types, 2009.

Intensity	*High (C)*	*Medium (C)*	*Medium (O)*	*Low (C)*	*Low (O)*
Farms (n)	134	135	22	134	22
Size (ha)	118	111	134	107	148
FBI (£/ha)	600	568	703	397	441
Input (£/ha)	353	302	163	280	131
NSO (tWe/ha)	16.9	12.7	12.8	9.8	6.5
FBI (£/tW_e)	36	45	55	40	68
Efficiency (kgW_e/£In)	47.8	41.9	78.5	35.1	49.4
GHG emissions (kgW_e/ $kgCO_{2e}$)	1.5	1.2	2.1	1.0	1.1

Key: C=conventional; O=organic; FBI=farm business income; Input=fossil-fuel based crop inputs (ferts, sprays, diesel) and water; NSO=net system output; We=wheat equivalent (excluding residual N); GHG=greenhouse gases; CO_{2e}: Carbon dioxide equivalent including methane, nitrous oxides.
Source: Lillywhite *et al.* (in progress).

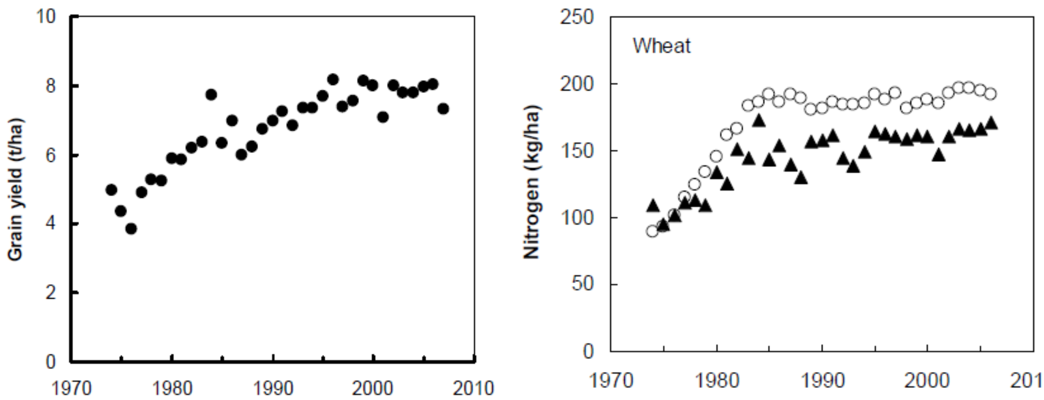


Figure 4-1: UK average wheat yields (●), N fertiliser use (O) and N removal (▲)
Source: Sylvester-Bradley *et al.* (2008)

The Ponti *et al.* (2012) study did not specifically consider nitrogen use intensity *per se* as an explanation. Seufert *et al.* (2012) did, but from the perspective that organic performance improved where N availability was higher, identifying N as a major yield-limiting factor in many organic systems. However, the better performance of legumes and perennials could be due to better N utilisation, rather than higher N levels. Ponisio *et al.* (2014) found that increased nitrogen use in conventional production widened the yield gap to organic.

The nitrogen issue could also explain why in many developing countries, resource-poor farmers unable to afford purchased N fertilisers have demonstrated potential to increase yields using organic/agroecological approaches (Scialabba and Hattam, 2002; Schutter, 2010; Badgley *et al.*, 2007). However, many of these studies are not of certified organic systems (certification is not relevant in self-sufficiency contexts) and in some cases involved non-permitted inputs and were therefore excluded from the Ponti *et al.* (2012) review. They also argued that in many cases the conventional yields were far below best practice and did not give a fair representation of the potential performance (an argument that can also be used for some of the organic data).

Seufert *et al.* (2012), in marked contrast to the earlier studies, found organic yields to be 43% lower than conventional in developing countries. However, the majority of these studies had atypical conventional yields more than 50% higher than local yield averages. They were not able to identify a single study meeting the selection criteria for their meta-analysis comparing organic and subsistence agriculture and highlighted the need for further research. Ponisio *et al.* (2014) found no significant difference in relative yield performance between organic and conventional crops in developing and developed countries.

4.2.4 Agroforestry

A central hypothesis in agroforestry is that productivity is higher in agroforestry systems compared to monocropping systems due to complementarity in resource-capture; i.e. trees acquire resources in space and time that the crops alone would not (Cannell *et al.*, 1996). This is based on the ecological theory of niche differentiation; different species obtain resources from different parts of the environment. Tree roots generally extend deeper than crop roots and so access soil nutrients and water unavailable to crops, as well as absorbing nutrients leached from the crop rhizosphere. These nutrients are then recycled via leaf fall onto the soil surface or fine root turnover. This should lead to greater nutrient capture and higher yields by the integrated tree-crop system compared to tree or crop monocultures (Sinclair *et al.*, 2000). Equally, the tree canopy occupies space above surface crops, making better use of above ground space for interception of sunlight and photosynthesis, with tree leaves continuing to harvest solar energy for longer periods than most annual crops.

The Land Equivalent Ratio (LER), first proposed by Mead and Willey (1980), is a means of comparing productivity of polycultures and monocropping systems. It is calculated as the ratio of the area needed under sole cropping to the area of intercropping at the same management level to obtain a particular yield:

$$\text{LER} = \frac{\text{(Tree agroforestry yield)}}{\text{(Tree monoculture yield)}} + \frac{\text{(Crop or livestock agroforestry yield)}}{\text{(Crop or livestock monoculture yield)}}$$

If a rotation includes more than one crop, a weighted ratio for each crop can be used, based on its proportion in the rotation. A LER of 1 indicates that there is no yield advantage of the intercrop compared to the monocrop, while, for example, a LER of 1.1 would indicate a 10% yield advantage. Under monocultures, 10% more land would be needed to match yields from intercropping (Dupraz and Newman, 1997). Yields can be expressed in physical units so that the LER refers to the biological efficiency of the mixture, or monetary units where the LER indicates the economic efficiency of the mixture. As a ratio, the result is independent of the yield units used. The LER reflects the ability of crops to partition resources in space and time, so that lower (physical) values of LER are recorded from mixtures of grasses in pasture, intermediate values from dissimilar vegetables, cereals and legumes, and highest values in agroforestry systems (Dupraz and Newman, 1997).

Werf *et al.* (2007) calculated Land Equivalent Ratios (LER) for two lowland poplar silvoarable trial systems in lowland England and found that LER values stayed above one for the 12 years after establishment. Newman (1986, in Dupraz and Newman, 1997) calculated LER values of 1.65 and 2.01 relating to economic and biomass yield respectively for a pear orchard/radish (*Raphanus sativus*) system. Dupraz (1994, in Dupraz and Newman, *op cit.*) modelled LERs for a *Prunus avium/Festuca arundinacea* system in France and estimated annual LERs of 1.6 in the early years after establishment, declining to 1.0 later in the rotation, with an average of 1.2 over the 60 year rotation.

Biophysical modelling of hypothetical silvoarable systems in Spain, France and the Netherlands using the YieldSAFE model predicted LERs of between 1 and 1.4, indicating higher productivity where crops and trees were integrated on the same land area compared to separate monocultures (Graves *et al.*, 2007).

In addition to higher yield potentials of agroforestry, product diversification may increase the potential for financial returns, by providing annual and periodic revenues from multiple outputs throughout the rotation and reducing the risks associated with farming single commodities (Benjamin *et al.*, 2000). Tree products can be used on the farm (e.g. for fence posts, fodder or bioenergy) and this, combined with greater resource-use efficiency (e.g. nutrient use), should reduce inputs and increase the 'eco-efficiency' of the farming system. However, the potential financial benefits may not be realised if suitable markets are unavailable, or if they are outweighed by additional labour and machinery costs. The establishment costs, as well as the time before the agroforestry component becomes productive, will also affect overall financial performance (see section 4.6.4).

4.2.5 Labour requirements

A common objection to many agroecological approaches is their perceived need for increased labour (Tripp (2005) cited by Pretty *et al.* (2011)). Labour has a cost that many farmers are keen to minimise, and increasing labour productivity is seen as an important driver for economic growth. Higher labour requirements may, therefore, be seen as a disadvantage. At the same time, generating employment for the rural economy is a key objective of rural development policy and also seen as one of the potential additional benefits of agroecology and organic farming (Lobley *et al.*, 2009), particularly if the financial returns generated also contribute to enhanced labour incomes or if opportunities for other members of the farm family are created. In a developing country context, this is very site specific – labour availability is a constraint in some areas (e.g. in areas strongly affected by HIV/AIDS), whereas in other areas where there are few alternatives, generating employment opportunities is welcome (Pretty *et al.*, 2011) This may also be true in UK and other European contexts. For example, Chatterton *et al.* (2015) highlight the potential employment generation benefits of UK livestock production, co-produced with environmental benefits in pastoral landscapes.

The question of production systems requiring higher labour inputs when chemical inputs are substituted by more labour-intensive practices has been particularly discussed in the context of organic farming. To our knowledge, there is very little robust research into the labour requirements of other types of agroecological farming in Europe[33]. This is likely, at least in part, to be due to the lack of a commonly accepted definition on which a comparison using existing data sources (e.g. the Farm Accounting Data Network (FADN)[34]) could be based.

Early European studies of labour use on organic farms (based on surveys and an evaluation of labour data in FADN type samples and reviewed by Lampkin (1994), Offermann and

[33] Search using Google Scholar with the search terms agroecology, labour, Europe
[34] http://ec.europa.eu/agriculture/rica/

Nieberg (2000) and Jansen (2000)) indicated that labour-use was higher on organic holdings, particularly on high value horticultural holdings and farms with direct marketing and on-farm processing activities. The overall increase in labour was of the order of 10-20% per holding, but it was not always the case that more labour was required for individual enterprises.

Morison *et al.* (2005), based on a survey of 1144 organic farms in the UK and the Republic of Ireland compared with results of the Farm Structure Survey, concluded that organic farms employ 35% more labour per farm than conventional farms, but conventional farms employ 80% more labour per hectare, because organic farms were found to be larger than conventional. The difference may be a factor of system comparability, but also reflects that systems with higher production levels and financial output require and can pay for more labour input. Lobley *et al.* (2005) surveyed organic holdings and captured data on employment and other labour characteristics as part of a study on the impact of organic farms on rural development and concluded that organic farming produces more employment than conventional farms, but organic farms are more likely to employ casual or part-time labour.

In contrast, the annual Organic Farm Incomes in England and Wales reports (most recently Moakes *et al.* (2013, 2014)[35]), where the data is differentiated by farm type and compared with comparable conventional farms using a clustering procedure, indicate much lower or no differences in total labour per farm or labour use per hectare for most farm types, raising the question whether the increased labour requirements are restricted to a limited set of specific circumstances. Although yield levels were lower on the organic farms (see section 4.2.3), the financial output generated per labour unit was also similar across the farm types, with slightly better performance on organic livestock farms and slightly worse performance on organic cropping farms, including horticulture where much larger differences in other financial parameters had been identified (see section 4.6.3).

It is also important to consider the nature of any jobs that are created, in particular skills, remuneration and permanence (Jansen, 2000). Unskilled farm labourers, seasonally-employed picking fruit and vegetables, might not qualify as desirable rural employment, but studies investigating this in greater detail remain rare.

Agroforestry also illustrates the dual character of labour becoming an obstacle to adoption as well as an opportunity for creating additional employment, although this is significantly influenced by the design of the system and the potential for mechanisation. Successful tropical agroforestry systems show that management of intercropped systems is often intensive with high manual labour input required. The high cost of manual labour in Europe is thought likely to lead to greater reliance on agrochemical and mechanical input, especially when unfavourable combinations of trees and crops are used (Eichhorn *et al.*, 2006). Within the UK and across parts of Northern Europe, there has been a decline in opportunities for manual employment in rural areas over the last 20 years. Doyle and Thomas (2000) suggest that even where agroforestry displaces traditional, grass-based livestock systems, job gains from the ‘forestry’ component of the system will compensate for any job losses from a reduction in livestock. Where the trees used in agroforestry produce annual products such as fruit and nuts, additional pruning and harvest employment may be created, although this may be casual and insecure rather than permanent employment. There may also be positive implications for local industries supplying inputs and processing outputs from both the agricultural and forestry components of the system.

[35] http://www.organicresearchcentre.com/?go=Research%20and%20development&page=Socio-economics%20and%20policy&i=projects.php&p_id=7

4.2.6 Factors affecting the productivity of agroecological approaches

The conclusion of this section is that the productivity of integrated systems can be similar to that of conventional, intensive systems. Organic farming produces lower yields, typically 20-30%, but more or less in certain cases depending on conventional N-use intensity. Agroforestry systems have the potential to generate higher yields overall (20-30% increases as measured by LER compared with growing trees and crops separately). Table 4-5 summarises the key factors impacting on productivity, acknowledging that there are potentially large differences between individual farms and locations. In some instances, the assessments presented in the table rely on own judgement only (for example NSO and integrated/agroforestry systems), as no studies have applied this method to these systems.

Table 4-5: Factors affecting the productivity of agroecological compared with intensive conventional systems

Output parameter	*Integrated*	*Organic*	*Agroforestry*	*Key factors*
Yields per ha	0	-	+	N fertiliser use, land use, crop protection effectiveness
Net system output (NSO)	0	-	+	Utilisation of crops, competition by livestock for foods suited to humans
Land equivalent ratio (LER)	0	-	+	Complexity of polycultures, complementarity of resource needs
Labour use efficiency	+	-/+	-/+	Mechanisation, rural employment goals, high value enterprises
Input use efficiency	+	++	++	Energy use for input manufacture, access to inputs, use of on-farm resources, in the agroforestry case also access to deeper soil nutrient reserves and better exploitation of solar energy

- = less than conventional, 0 = similar to conventional, + = higher than conventional

Source: Own assessment based on literature presented in this section.

The results reviewed indicate that the intensive use of inputs, in particular nitrogen fertiliser and fossil energy, continues to play an important role in maintaining production levels with respect to land. However, if this is at the expense of using up non-renewable resources such as phosphates and fossil energy (see next section), then high production based on continued intensive use of inputs is not necessarily sustainable. Even if efficiency can be improved, so that fewer inputs are needed per kg food produced, further significant increases in production may still lead to an increase in the total use of non-renewable inputs.

However, if the agroecological alternatives involve greater system complexity, often leading to greater labour and management requirements, there is a need to consider productivity in terms of labour use related to physical as well as financial output. An overall indicator of productivity (such as Total Factor Productivity) would help ensure that the advantages of greater biological/land productivity are fully accounted for in terms of the other inputs used.

Opportunities to increase productivity through better system design and management, and potentially through the application of new technologies, have been identified for all three systems. They all demonstrate potential to improve the efficiency with which inputs are utilised in agricultural production. However, there is a lack of information about exactly where to apply each approach in order to get the best result.

More radical solutions, such as closing nutrient cycles and returning nutrients from cities to farmland, and focusing on systems that maximise solar energy capture rather than fossil energy use, may well be required. Concentrating livestock production in areas where they are complementary to, rather than competitive with, human food needs, for example by utilising pasture lands not suitable for crop production, may also help.

4.3 Energy use and greenhouse gas emissions

4.3.1 Assessing energy use and greenhouse gas performance

Despite the importance of solar energy for plant production, the energy used in agriculture essentially comes from direct, on-farm consumption of fossil energy (e.g. fuel and oil), as well as indirect energy consumption resulting from the production and transport of imported goods. The intensive process of producing nitrogen-based fertilisers represents the most energy expensive input for modern farming, accounting for about half of agriculture's energy use and approximately 1.1% of energy use globally (Dawson, 2011; Foresight, 2011).

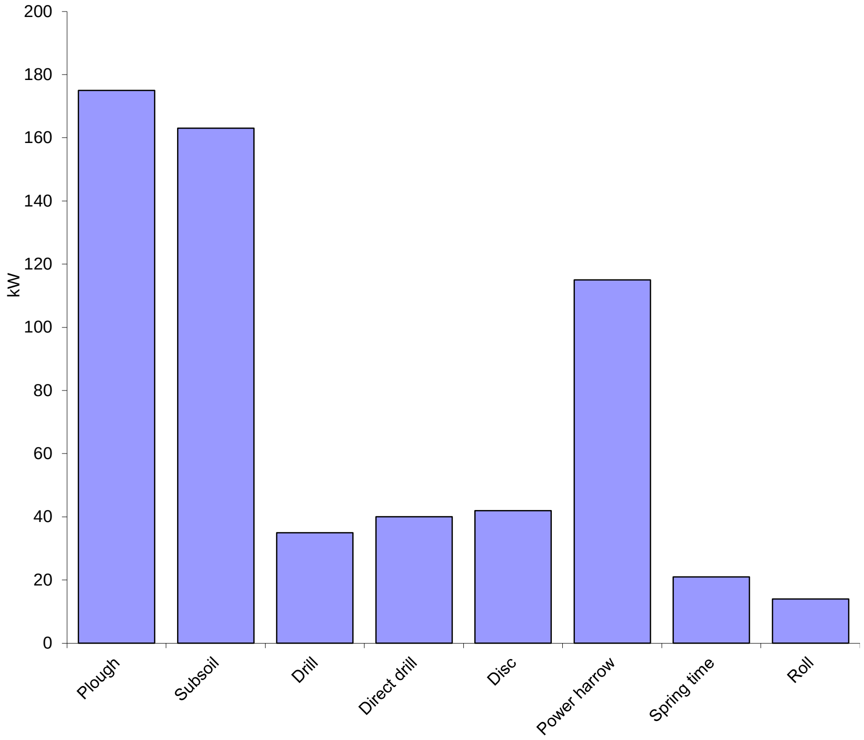


Figure 4-2: Energy used by machines (kW ha^{-1}) in different field operations
Source: Leake (1997)

In addition, global agriculture is responsible for about 10–12% of greenhouse gas emissions; and this figure increases to over 30% when emissions associated with land use change (from forestry to crop production or the cultivation of peatlands) are included (Foresight, 2011). In light of the reduced availability of fossil fuels, and the effects of climate change, assessing the impact of agricultural systems on greenhouse gases and energy use is critical to determine their respective contribution to sustainable intensification.

The energy efficiency of agricultural systems is commonly assessed by converting all inputs to a common metric (megajoules - MJ). Some studies have used kilocalories (kcal) as a unit of comparison, and/or have included the embodied energy in the food produced to compare the relative inputs and outputs (Pimentel *et al.*, 1983).

The most common metric used for assessing greenhouse gas emissions from agricultural systems is the Global Warming Potential (GWP) expressed as the sum total of greenhouse gases in kilograms of CO_2 equivalent (CO_2e) over a 100 year time frame. On a global basis, nitrous oxide (N_2O) emissions from soils, and methane (CH_4) from enteric fermentation of ruminants are the largest contributors to agriculture's GWP (excluding land use change). These gases have a GWP 298 and 25 times stronger than CO_2 respectively, so their reduction is emphasised within agricultural policies and industry roadmaps.

The total GWP and/or energy use of contrasting practices and farm systems is usually expressed either per unit of product or per hectare of land. Assessments can include all energy use and emissions within the various production stages of an agricultural product through the application of Life Cycle Assessment (LCA) or the emissions and energy used within the farm only. Other assessments include only certain inputs to the farm that account for a large percentage of the total impact (e.g. fertiliser or feed (Smith and Little, 2013)). Some studies have also taken the dimensionless 'energy ratio' approach to determine the efficiency of production systems (e.g. dividing the energy output in food sold by the energy input of fossil fuels and/or the associated GWP). Lampkin (2007) highlights that this method can be a useful determinant of the efficiency of agricultural systems in capturing solar energy and transforming this into feedstuffs for growing populations. Halberg *et al.* (2005) also highlight the potential of this approach to allow farmers and advisors to compare the efficiency and environmental impacts of crop and livestock enterprises, in order to identify areas for improvement. The use of proxies for measuring farm performance is also becoming more widespread, in view of the detailed data requirements of LCA. For example, the Organic Research Centre's Public Goods Tool (Gerrard *et al.*, 2012) and the Linking Environment and Farming (LEAF) audit ask the farmer if they have implemented key practices that can increase/decrease the farm's greenhouse gas footprint (e.g. any major land use change within a 20 year period or covering manure stores). This approach seeks to overcome the sometimes extensive time commitment required for a more quantitative assessment, while still providing a meaningful overview of a farm's performance.

Agroecological practices and systems can contribute to greater energy efficiency and lower greenhouse gas emissions from agriculture, not only through reduced cultivations and other forms of direct use, but also through reduced usage of synthetic mineral fertilisers, pesticides and other inputs which depend on fossil energy for their manufacture. The potential of these practices to contribute towards a more energy and greenhouse gas efficient agriculture is explored in more detail below.

4.3.2 Integrated crop/farm management

Integrated farming techniques have the potential to reduce energy consumption, particularly through a reduction in pesticide and fertiliser inputs and through the adoption of reduced/zero tillage techniques. However, the continued reliance on synthetic nitrogen fertiliser in preference to biological fixation by legumes means that the full potential for fossil energy reductions may not be realised.

Reduced/zero tillage techniques have been shown to significantly reduce energy requirements for cultivations (see also sections 3.2.2. and 3.6.2). For example, a study by Michigan State University found a lower fuel use for a corn, soybean, and wheat rotation under conventional no till, compared to the same rotation under organic and low-input conditions (Robertson *et al.*, 2000). Other research has shown that the use of low-input production combined with reduced tillage can lead to substantial improvements in energy efficiency, even when the embodied energy associated with inputs is accounted for (Clements *et al.* 1995).

Many studies have shown that increased soil organic matter provides the opportunity for carbon sequestration. However, studies on reduced and zero-tillage have tended to focus on the top 10 cm of soil and compared this to the ploughed equivalent. Longhurst (2010) showed that while zero tilled soils did retain greater levels of carbon in the upper horizon, when sampling depth was increased to 20 cm the ploughed system showed equivalence across all horizons. A Defra Scientific Report compiled by Bhogal *et al.* (2006) on the carbon content of arable soils in England concluded the following:

- Many of the increases in soil organic carbon (SOC) measured following reduced/ zero tillage have been confined to the top 10-15 cm. Where deeper soil samples have been taken, apparent differences between tillage systems have often disappeared;
- There has only been a limited number of contrasting tillage studies in the UK, with most looking at zero tillage rather than reduced tillage practices;
- The best estimate of the C storage potential of zero tillage under English and Welsh conditions is 310 (± 180) kg C ha^{-1} yr^{-1}, based on measurements at six study sites. This equates to *ca.* 0.35% of the typical carbon content of an arable soil in England and Wales;
- Reduced tillage is estimated to have half the C storage potential of zero tillage at 160 kg C ha^{-1} yr^{-1};
- These estimated C storage potentials can only be regarded as the initial rate of increase (< *ca.* 20 years). Annual rates of SOC accumulation decline (eventually to zero) as a new equilibrium is reached (after > *ca.*100 years);
- SOC accumulation is finite and reversible. SOC levels will only remain elevated if the practice is continued indefinitely. These estimates of potential C storage from zero and reduced tillage should therefore not be considered to be annually cumulative, as in the UK, tillage land is ploughed every 3 to 4 years to reduce the build-up in weeds, disease and soil compaction levels. It is arguable that much (if not most) of the stored C will subsequently be released as a result of the increased soil disturbance caused by ploughing. However, further work is required to establish what effect periodic ploughing would have on long-term SOC accumulation in such systems.

4.3.3 Organic farming

Energy use

In general, organic farms use less energy than conventional farms, at least on a per hectare basis (Figure 4-3). There are several factors that contribute to this.

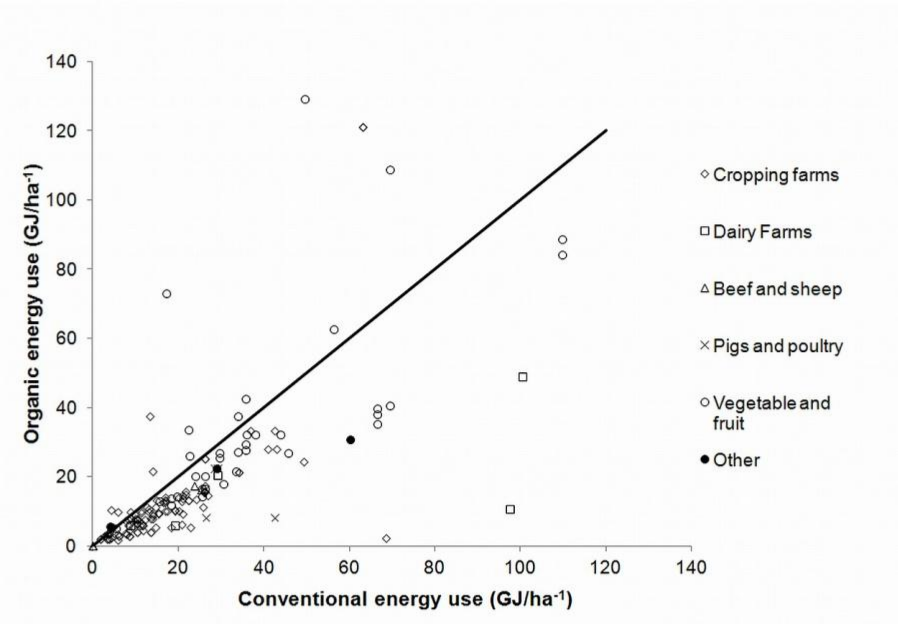


Figure 4-3: Comparison of the energy efficiency of organic and non-organic production systems. Source: Smith *et al.* (2014b)

Within the organic sector, the EU Organic Regulation prohibits synthetic nitrogen fertiliser and the use of most pesticides and the non-use of these inputs is a major contributing factor to a lower energy requirement in organic cropping (Stolze *et al.*, 2000; Lampkin, 2007). For certain crops, however, (e.g. potatoes, broccoli, lettuce), organic methods can offset the reduction in man-made chemical inputs by increased mechanical labour (Pimentel *et al.*, 1983; Williams *et al.*, 2006; Venkat, 2012). Cormack and Metcalfe (2000) also found that the lower yield and the inclusion of fertility building crops within an organic stockless arable farm resulted in a lower energy efficiency overall. Crops with high rates of loss from pest and disease and/or high cosmetic standards (e.g. potatoes and apples) can also be less efficient, due to reduced marketable yields. Organic tomatoes and other greenhouse crops have also been found to have lower energy efficiency due to similar levels of inputs as conventional management and a reduced yield under organic conditions (Williams *et al.*, 2006; Alonso and Guzman, 2010).

Systems using composts and manures also compare less favourably to intensive, conventional systems when the energy content of the organic matter/compost used is taken into account. Average energy inputs per unit of land area were approximately double that of the conventional farms when this was included within system comparisons by Karlen *et al.* (1995). However, Alonso and Guzman (2010) point out that inputs of manure and compost help to promote the long-term health of the system, and as such cannot be compared in the same way to non-renewable energy sources. Low-input conventional systems can also be highly energy efficient when the use of pesticides and tillage is tightly controlled (Clements *et al.*, 1995; Robertson *et al.*, 2000; Snyder and Spaner, 2010; Zentner *et al.*, 2004).

For livestock systems, recent reviews have found that most grazing systems adopting agroecological practices will require less fossil energy on a unit area or weight of product basis (Gomiero *et al.*, 2008; Smith *et al.*, 2014b). This is a direct result of the use of clover and other forage legumes within leys, which results in more efficient forage production compared to conventional practice (Hoeppner *et al.*, 2006; Deike *et al.*, 2008; Küstermann *et al.*, 2008). Similarly, for dairy systems, a reduced reliance on imported concentrates can leads to greater efficiency overall (Cederberg and Mattsson, 2000; Haas *et al.*, 2001; Thomassen *et al.*, 2008). The impact of purchased fodder on the energy use of different organic and conventional farm types in Switzerland can be seen from the result of survey data presented in Figure 4-4, with mixed farms in particular showing large differences (Schader, 2009).

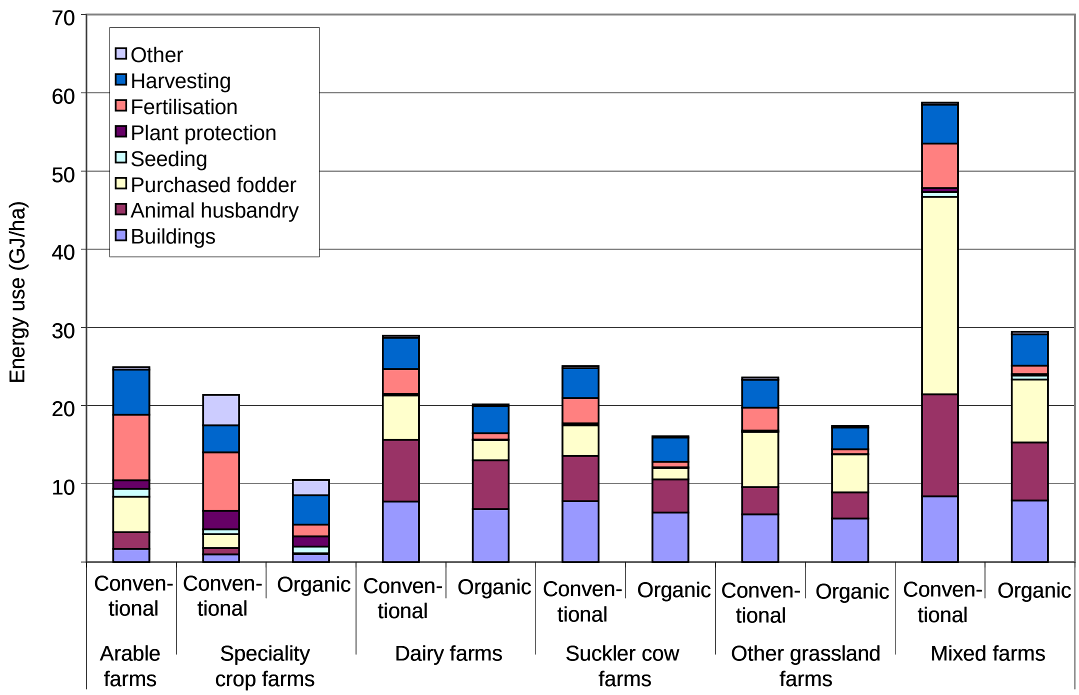


Figure 4-4: Energy use on organic and conventional farms in Switzerland, by type
Source: Schader (2009)

With regard to poultry, both organic meat and egg production tend to require more energy per kg product and per ha under less intensive management regimes, as poorer overall feed conversion ratios and higher mortality rates can reduce overall efficiency (Williams *et al.*, 2006; Leinonen *et al.*, 2012). Performance will vary greatly according to the range composition, however. The extent to which foraging can contribute to the nutritional needs of pigs and poultry is being investigated, with early results suggesting that pasture systems incorporating alfalfa can considerably reduce reliance on soy-based feed within organic pig production (Jakobsen, 2014), although longer life-spans and the lower growth rates associated with more extensive forage-based production can result in systems with a greater impact on energy use (EBLEX, 2009; Gunnarsson *et al.*, 2011; Ross *et al.*, 2014).

<u>Greenhouse gas emissions</u>

In general terms, the reduced use of fossil energy in organic farming will contribute to reduced CO_2 emissions, at least on a per ha basis. The reliance on legumes as the main source of N in organic systems also avoids the N_2O emissions associated with mineral nitrogen fertiliser manufacture and application (El-Hage Scialabba and Müller-Lindenlauf, 2010). The increasing popularity in the use of legumes on non-organic farms can result in savings for the agriculture sector overall (Defra, 2013a). The greater use of mixed crop and livestock systems on organic farms allows for the cooperative use of farmyard manures between enterprises, with potential to reduce emissions compared with more specialised systems (Niggli *et al.*, 2009). The use of legumes and livestock manures in organic cropping systems can also lead to greater amounts of soil carbon, both through direct additions and through reducing the breakdown of soil organic matter by soil organisms requiring energy to make use of easily available nutrients (Niggli *et al.*, 2009; Olesen, 2009; Smith *et al.*, 2011).

A recent meta-analysis of 74 studies conducted by Gattinger *et al.* (2012) confirms higher soil organic carbon concentrations (0.18 ± 0.06%) and stocks (3.50 ± 1.08 Mg C ha^{-1}) in top soils under organic management. When the analysis within this study was restricted to organic systems with zero net inputs and retaining only the datasets with highest data quality, the mean difference in soil organic carbon stocks between the farming systems was still significant (1.98 ± 1.50 Mg C ha^{-1}). It is likely that these benefits will be greatest where a fertility-building (N and C fixing) phase involving grass/legume leys or green manures is introduced into exploitive arable/horticultural cropping sequences, as these crops can compensate for the use of plough-based tillage and cultivations for weed control in the absence of herbicides.

Leifeld (2013) criticised the study by Gattinger *et al.* (*op cit.*), highlighting that, in many of the studies, the amount of organic fertiliser (manure and/or compost) in the organic systems exceeded the amount applied in the conventional. Leifeld and Fuhrer (2010) argue that a truly unbiased comparison of management types should be based on similar organic fertiliser rates and crop rotations incorporating fertility building leys, as neither of these aspects are unique to organic farming. While this is true, an experiment of this kind would lose the significance of the farming system. In reality, all practices used by organic farmers can be used by other farmers and are not unique, but organic farmers are more likely to be using a fertility-building period in their crop rotation and organic manures than non-organic producers. European organic regulations (EC, 2008) dictate that the fertility of the soil should be maintained and increased through crop rotations including legumes, and through application of manures or other organic material. Certification bodies, such as the Soil Association in the UK, also require certified producers to include a balance of cropping and grass/clover leys in their crop rotations (Soil Association, 2008).

Reduced GHG emissions per ha on organic farms do not necessarily translate to reduced emissions per unit of food produced as a result of the lower yields on organic farms (see section 4.2.3). A recent literature review of Life Cycle Assessment (LCA)-based studies compared the total Global Warming Potential (GWP) of organic and conventional products (Knudsen *et al.*, 2011; Figure 4-5).

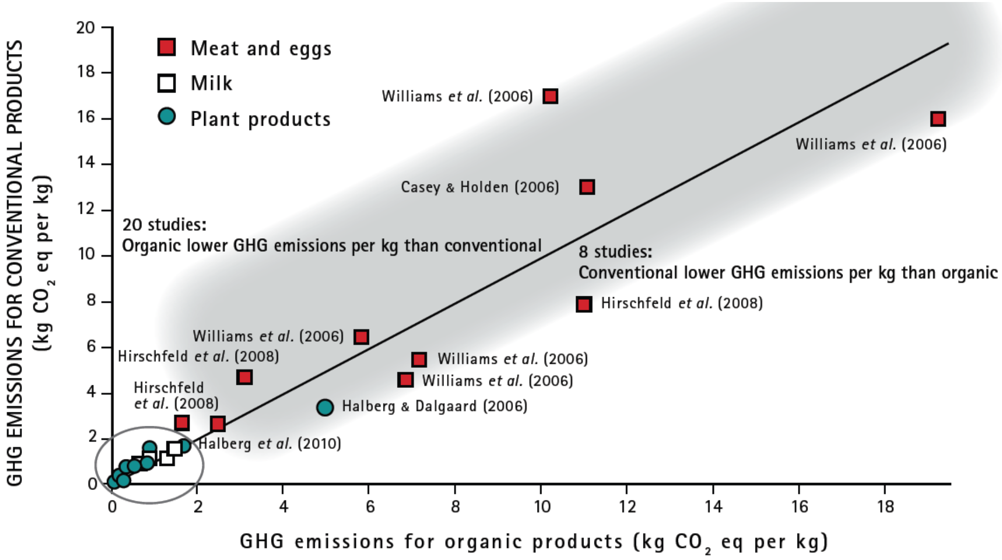


Figure 4-5: Literature review comparison of greenhouse gas emissions for conventional and organic products.
Organic performs better above the line, worse below the line
Source: Knudsen *et al.* (2011). Reproduced with the permission of the Food and Agriculture Organisation of the United Nations.

This study confirmed that there is no significant difference overall when comparisons are made per unit of product, with the effect of lower yields under organic management being offset by lower inputs. GHG emissions per kg are much lower for plant than for animal products, and the variability in results for plant products is also low, consistent with the results presented in Table 4-3 above. The values for milk are lower than for other livestock products, but this may be related to the low dry matter content of the product. For meat and eggs, the variation between studies for the same product types is considerable, which reflects the more complex range of factors affecting potential emissions.

In the case of ruminant livestock, reduced reliance on imported feed within organic systems can help to avoid deforestation/land clearance for growing crops such as soya and maize (Cederberg and Mattsson, 2000; Jørgensen *et al.*, 2005; Thomassen *et al.*, 2008). Domestically-produced grass/clover also has less impact on greenhouse gas emissions than conventional forage, due to reduced use of synthetic fertilisers (Dalgaard *et al.*, 2001). Increased longevity within organic cattle systems can reduce the relative emissions from the unproductive rearing of dairy and beef cows (Lynch *et al.*, 2011). Increased milk yields can also lead to a decrease in animal fertility and health and to an increase in the overall replacement rate (Novak and Fiorelli, 2009). Greenhouse gas emissions can be reduced further if dairy calves produced are used for beef production, reducing or offsetting the need for suckler cows (Flysjö *et al.,* 2012). This study highlights the issue of system boundaries and rules of allocation when allocating emissions to final products, but the results may not be transferable to the UK as there is only limited use of dairy calves for organic beef production due to market preference.

However, when considering the overall impact on greenhouse gases per unit of animal product, stocking rates, growth rates and/or milk/egg yields per head, as well as increased maintenance feed intakes for free range pigs and poultry, all need to be considered. Lower stocking rates on organic farms can result in lower methane (CH_4) emissions per ha, as a high proportion of emissions are animal-related (Schader *et al.*, 2012). However, increasing yields per animal can decrease emissions per unit of product (Lovett *et al.*, 2005; Lovett *et al.*, 2006; Garwes, 2009; Zehetmeier *et al.*, 2012). Research in Scotland has shown that permanent housing of dairy cattle in low-forage, zero-grazing systems, which are not permitted on organic farms on animal welfare grounds, can reduce GHG emissions by up to

25% (Ross *et al.*, 2014). Land use change impacts due to increased soya use, and embodied energy in bought-in feedstuffs, may be relevant in such contexts (see above).

Milk yields are estimated to be 10-20% lower per cow, and even lower per hectare, on organic dairy farms (see for example Table 4.2 above), potentially resulting in higher methane emissions per kg of milk produced (Piorr and Werner (1998) in Stolze *et al.* (2000)). Increasing the roughage content of the diet (a common practice on organic farms) can also result in an increase in methane emissions (Boer, 2003), although there is some evidence that diets high in tannins (e.g. diets with a high clover/legume content) may produce less methane than grass-only diets through a suppression of fibre degradation in the rumen (Hess *et al.*, 2006). Cederberg and Mattson (2000) found that reductions in the nitrous oxide (N_2O) emissions associated with the manufacture of synthetic fertilisers (used for grass and concentrate feed production in conventional systems) more than offset the greater amounts of methane per litre released by organic dairy cattle (Figure 4-6). Similar results were found by Allen *et al.* (2007) for UK dairy farms (Table 4-6), which also show the potential for the best in each group to outperform the average.

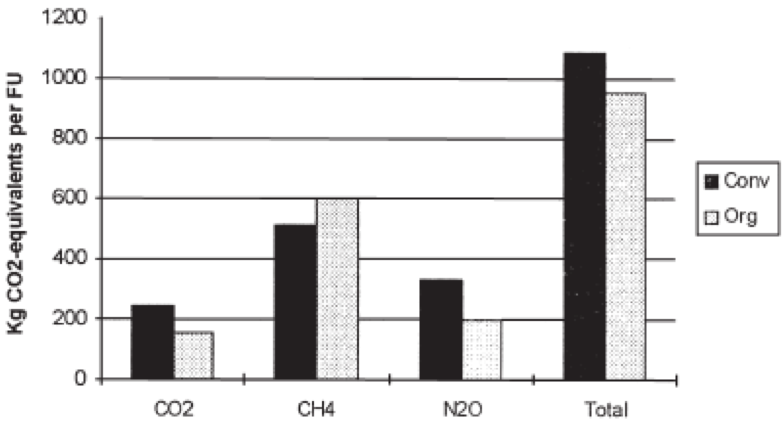


Figure 4-6: Global Warming Potential per functional unit (FU = t milk) from contrasting production systems
Source: Cederberg and Mattsson (2000)

Table 4-6: Combined greenhouse gas emissions from UK organic and conventional milk production

System	***Conv. average***	***Conv. top 25%***	***Org. average***	***Org. top 25%***
g CO_{2e} per litre milk	907	745	828	705
% from CO_2	23	25	21	22
% from CH_4	52	55	69	68
% from N_2O	25	20	10	10

Source: Allen *et al.* (2007)

4.3.4 Agroforestry

Combined food and energy systems, incorporating crops, livestock and energy crops such as willow coppice, can compare favourably in term of energy use to conventional modes of production (Reith and Guidry, 2003; Ghaley and Porter, 2013). There has also been considerable interest over the last 20 years in investigating the potential of agroforestry as a tool for addressing the issues of climate change through mitigation and adaptation (Adger *et* *al.*, 1992; Schroeder, 1994; Albrecht and Kandji, 2003; King *et al.*, 2004; Lal, 2004; Montagnini and Nair, 2004; Peichl *et al.*, 2006; Schoeneberger, 2009).

Agroforestry has the potential to contribute by increasing afforestation of agricultural lands, by reducing resource use pressure on existing forests and by producing both durable wood products and renewable energy resources (Dixon, 1995). Agroforestry can increase the amount of carbon sequestered compared to monocultures of crops or pasture due to the incorporation of trees and shrubs (Jose, 2009). Woody perennials store a significant amount of carbon in above ground biomass and also contribute to below ground carbon sequestration in soils. The potential for agroforestry systems to sequester carbon depends on a number of factors including system design, tree density per unit area, species composition and age, environmental factors such as climate, management and the end product. Schroeder (1994) estimated average carbon storage by agroforestry systems as 9, 21, 50 and 63 Mg C ha^{-1} in semiarid, subhumid, humid and temperate regions, with higher rates in temperate regions reflecting longer rotations and longer-term storage.

Sharrow and Ismail (2004) found that a Douglas-fir (*Pseudotsuga menziesii*)/ perennial ryegrass (*Lolium perenne*)/ subclover (*Trifolium subterraneum*) silvopastoral system in Oregon, USA, was more efficient at storing C than tree plantations or pasture monocultures. In the 11 years since establishment, the silvopastoral system had accumulated 740 and 520 kg ha^{-1} year^{-1} more C than forests and pastures respectively. They suggested that this was a result of higher biomass production and active nutrient cycling patterns within the silvopasture system compared to tree and pasture monocultures. Peichl *et al.* (2006) also recorded larger total C pools in poplar (*Populus*)/barley and spruce (*Picea*)/barley agroforestry systems compared to a barley monocrop (96.5, 75.3 and 68.5 Mg C ha^{-1} respectively). Gupta *et al.* (2009) observed increases in soil organic carbon, from 0.36% in monocropped cereals to 0.66% in poplar/cereal agroforestry soils, amounting to 2.9-4.8 Mg ha^{-1} more soil organic carbon in agroforestry soils.

In a study of carbon sequestration potential in a tree-based intercropping system in Guelph, Ontario, the permanent tree component (13 year old hybrid poplars) sequestered 14 Mg C ha^{-1} year^{-1}, and C contribution from leaf litter and fine root turnover was estimated at 25 Mg ha^{-1} year^{-1}; over the 13 year period this amounts to the immobilisation of 156 Mg ha^{-1} (Thevathasan and Gordon, 2004). Taking into account the release of C back into the atmosphere as leaf litter and fine roots decompose, the net sequestration potential of trees was calculated as 1.65 Mg C ha^{-1} year^{-1} or approximately 7 Mg ha^{-1} year^{-1} of CO_2 (Thevathasan and Gordon, 2004).

Ammonia (NH_3), although not a greenhouse gas, can result in damage to sensitive plants and soil ecosystems as well as to human health. In the UK, agricultural production accounts for over 80% of NH_3 emissions, which come from livestock housing, grazing, and storage and spreading of manure (Misselbrook *et al.*, 2010). Trees are effective scavengers of both gaseous and particulate pollutants from the atmosphere, suggesting that increasing tree cover within agricultural landscapes can remove NH_3 from the atmosphere near the source, thereby reducing impacts on sensitive ecosystems. A recent project, *Agroforestry systems for ammonia abatement* (Defra project AC0201) running from 2007-2011, aimed to quantify the emission abatement of agricultural ammonia (NH_3) that is achievable with a range of different on-farm woodland features including downwind shelterbelts, silvopastoral systems and wind breaks, at the UK scale (Bealey *et al.*, 2013). The project included experimental work in a wind tunnel facility and in the field, as well as modelling simulations. Wind tunnel experiments showed that significant ammonia can be recaptured by trees, with the source height the key factor in determining the effectiveness of tree belts as a mitigation measure (Bealey *et al.*, *op cit.*). Modelling of NH_3 capture by shelterbelts and understorey scenarios predicted maximum deposition rates of 28% for shelterbelts around a housing source and 60% for understorey (e.g. woodland chicken) sources. In field case studies of NH_3 concentrations downwind of poultry houses with and without trees on the Food Animal

Initiative (FAI, www.faifarms.com) farm near Oxford, concentrations downwind of wooded transects were 10-25% lower than the unwooded transects (Bealey *et al.*, *op cit.*).

4.3.5 Factors affecting the energy use and greenhouse gas emissions of agroecological approaches

In summary (see Table 4-7), the use of agroecological approaches can often, but not always, improve energy efficiency and reduce greenhouse gas emissions through:

- Reduced inputs with greater reliance on renewable/biological sources within the farm (e.g. N fixed by legumes) or home-grown sources of feed that do not contribute to deforestation. In particular, using legumes for N supply can improve the efficiency of grazing livestock systems using grass/clover leys;
- Mixed systems providing opportunities for complementarity between different enterprises, reducing the need for external inputs and/or nutrient surpluses;
- Increasing carbon sequestration by using methods that can build soil carbon (e.g. application of manures and the use of grass clover leys in the rotation) as well as the use of woody perennials in agroforestry systems.

Table 4-7: Factors affecting the energy use and greenhouse gas emissions of agroecological compared with intensive conventional systems

Output parameter	***Integrated***	***Organic***	***Agroforestry***	***Key factors***
Energy use for cultivations	-	+/-	-	Reduced/zero tillage techniques reduce energy use, but non-use of herbicides in organic can lead to increased use
Energy use for other inputs	0/-	--	-	Use of biological N-fixation and alternatives to biocides reduce energy for input manufacture
Soil organic carbon	0/+	+	++	Fertility building phase in organic rotations and agroforestry treelines, possible impact of reduced tillage
Above ground carbon sequestration	0	+	++	Increased grass/legume component on organic farms and trees in agroforestry
GHG emissions per ha per unit product	 - -	 -- 0/+	 -- --	Link to sequestration, lower energy and other external input use, unit product values affected by yield.

- = less than conventional, 0 = similar to conventional, + = higher than conventional

Source: Own assessment based on literature presented in this section.

The denominator (per ha or per unit product) will affect the relative performance of farms implementing the measures outlined above. In particular, in the case of organic farming, the differences are reduced and may even be reversed per unit of production, with the gains from a reduction in inputs and other practices offset by reduced yields and potentially increased cultivations in the absence of herbicide use.

Free range pig, poultry and other systems may also be adversely affected by higher maintenance feed requirements compared to conventional fully-housed systems, but permanent housing may be less acceptable to consumers on welfare grounds. Land use change due to the production of feed concentrates (e.g. deforestation for soya production) may also be relevant in this context.

While some evidence suggests that methane emissions per unit product can be reduced by permanent housing and intensification, this does not preclude improved and comparable performance from low-input systems, particularly if other greenhouse gases are considered alongside CH_4.

4.4 Biodiversity and related ecosystem services

Biodiversity can be considered at the genetic, species and ecosystem level. It is of interest to human populations for both intrinsic (i.e. cultural, social, aesthetic and ethical benefits) and instrumental (i.e. directly used for food, fuel, recreation, or indirectly via ecosystem processes and environmental services) reasons (Decaens *et al.*, 2006).

While biodiversity is of important *per se*, and a major focus of nature conservation and agri-environmental policies as a consequence, it is also of relevance in terms of the support services that it provides to agriculture, not least because agriculture is fundamentally concerned with the provisioning ecosystem services of food and energy production. However, as highlighted elsewhere in this report, biodiversity also contributes to system self-regulation, in particular of pests, parasites and diseases, as well as to pollination, enabling plant reproduction, genetic resources and soil health and water quality. Biodiversity loss as a consequence of agricultural intensification has had a negative impact on these ecosystem services (NEA, 2011). To an extent they can be replaced by other inputs, but at a cost.

Pollinators in particular are a cause for concern. Approximately 250 crop species are grown in Europe of which 150 are believed to be pollinated by insects and the global value of pollinators in 2005 was estimated to be €120 billion (Gallai *et al.,* 2009). It is therefore a concern what impact the decline in farmland pollinators, particularly honeybees and bumble bees (Biesmeijer *et al.,* 2006; Potts *et al.*, 2010), as a result of agricultural intensification (Goulson, 2003), will have on national and global food production. Pollinators are now a major focus for the National Pollinator Strategy for England (Defra, 2014b) and the English Countryside Stewardship pollinator package available from 2016 (Defra, 2015).

In this section, the impacts of different agroecological approaches on biodiversity and related (non-provisioning) ecosystems services are assessed.

4.4.1 Assessing biodiversity performance

Biodiversity is a critical performance indicator for sustainable intensification, but there are a number of inherent difficulties in comparing one system with another. These include:

- the basis of comparison, including the per unit area or unit product argument rehearsed elsewhere in this report, but also what type of farms should be compared when different farming systems have inherent and immutable differences;
- the mobility of certain elements of biodiversity, e.g. birds and bats, and the impact of, and interaction with, the surrounding landscapes (Gabriel *et al.,* 2010; Hodgson *et al.,* 2010);
- the biodiversity indicators used, given that directly measuring biodiversity can entail very high costs, due to the wide range of different components and their temporal variability.

The usual parameter for assessing biodiversity is species richness (i.e. number of species), with evenness or abundance an important second parameter (Magurran, 2004). Both contribute to species composition of the community. Some studies have focused on endangered (red list) species, particularly birds and some plants.

From a functional biodiversity perspective, which is highly relevant to agroecosystems, abundance and spatial distribution of key species (earthworms, predators, pollinators) is of particular importance. Grouping species by function is becoming more widespread, but more evidence is still needed about their impacts on pollination, pest control, soil functioning etc. Species diversity is often used as a proxy for the services they provide, based on the assumption that greater diversity supports higher service provision. For example, higher pollinator diversity has been linked with higher crop yields (Hoehn *et al.*, 2008) as well as increasing the stability of production in a variable climate (Brittain *et al.*, 2013). In other studies, however, there is evidence that species identity is more important to ecosystem functioning than total species richness (Walker, 1992). Ecosystem services such as pest control and pollination can also be measured directly (Boatman *et al.*, 2010; Crowder *et al.*, 2010; MacFadyen *et al.*, 2009a; MacFadyen *et al.*, 2009b). Quantifying the economic values of biodiversity-mediated ecosystem services is a particular challenge, which is generally carried out by modelling approaches (e.g. Breeze *et al.*, 2011).

4.4.2 Integrated crop/farm management

Integrated systems mainly rely on maintenance and enhancement of native species and their habitats to allow full integration of natural regulatory processes, on-farm alternatives and management skills, in order to achieve maximum replacement of off-farm inputs, maintain species and landscape diversity, minimise losses and pollution, provide a safe and wholesome food supply and sustain income. Ecological infrastructure manipulation by enhancement of field margins provides farmland area for ecological compensation within government supported schemes and has been one of the key attributes of integrated farming. This has enhanced biodiversity and species richness, provided a habitat and food supply for predators of key pests, minimised ingress of problem weeds into field crops and losses (soil/overland flow) from farmland to controlled waters (Boatman, 2010).

A number of studies have shown benefits to biodiversity and wildlife through the adoption of non-inversion tillage. Changes in soil biota, both micro and macro, are positive as are the numbers of ground dwelling invertebrates. However, a study of carabid beetles in zero-tilled versus ploughed fields recorded almost four times as many individuals in the ploughed treatments compared to the zero-till. This is likely because the high level of surface trash restricts the movement compared to bare ploughed land as pitfall traps not only measure density but also activity. The Fisher Alpha diversity index of species assemblages showed the zero-tilled system to be significantly higher, indicating a more stable ecosystem (Longhurst, 2010).

Studies of birds visiting split field plots in winter showed a very high preference for zero-tilled stubbles sown with winter wheat over their ploughed comparison, particularly through the late winter period (Saunders *et al.*, 2000). The absence of food during this period is well known to be a major contributor to the decline in farmland birds in modern times. Tillage systems which retain resources close to the soil surface are more likely to be attractive to foraging birds.

A comprehensive review of Europe-wide evidence on the biodiversity impacts of reduced cultivations is provided by Holland (2004).

The 1998 Integrated Arable Crop Production Alliance (IACPA) Integrated Farming Report identified a number of environmental benefits delivered by IFM.

- IFM resulted in a more diversified farm crop mosaic, encouraging biodiversity and a more varied landscape;
- In general, at all sites, more habitats were created and hedges and field margins managed to encourage beneficial invertebrates. The full impact of this is not known although there is evidence of greater food availability for birds and one site clearly demonstrated higher levels of biological control and reduced insecticide use in cereal crops as a result;
- Integrated farms typically set-aside between 4 and 11% of land for the creation of semi-natural habitats;
- Increases were recorded in the numbers of bird species which showed a strong preference for integrated direct drilled stubbles over 'conventional' ploughed fields.
- Earthworm numbers increase where tillage is reduced in the integrated system.

While not displaying direct obvious benefits to yields, such as those associated with beetle banks and conservation, many non-cropped conservation margins around arable fields can be seen as a valuable offset for the recognised low biodiversity in field centres where herbicides are used. Margins sown with wild bird seed mixes could be considered an offset of this type, which are proven to provide vital forage opportunities for graniverous farmland birds during the winter period when other food availability is limited (Hancock and Wilson, 2003; Stoate *et al.*, 2003). Crops grown for game cover for example have been shown to support up to 100 times more farmland birds and significantly more species than surrounding conventional arable habitats (Parish and Sotherton, 2004). Agri-environmental scheme options have also been developed to reverse the decline of pollinator numbers. Agricultural pollen and nectar mixes sown into field margins have been shown to significantly increase the number of bumblebees (Carvell *et al.,* 2006; Pywell *et al.*, 2006; Carvell *et al.*, 2007). However the uptake of pollinator-supporting options such as pollen and nectar-sown margins has been relatively low.

4.4.3 Organic farming

The most up-to-date review comparing organic versus conventional farming and their biodiversity and environmental impacts conducted an hierarchical meta-analysis of 94 studies, using species richness of arthropods, microbes, birds and plants as the measure of biodiversity (Tuck *et al.*, 2014). This study improved on previous reviews by addressing the hierarchical structure of multiple within-publication effects sizes and including standardised measures of land-use intensity and heterogeneity across all studies. The meta-analysis found that on average, organic farming increased species richness by about 30% (Figure 4-7). This result was robust over the last 30 years of published studies, lending support for the argument that organic farming is a reliable method for increasing biodiversity on farmland and may help to reverse the declines of formerly common species. This effect was also robust across sampling scales, in contrast to other studies that suggest the benefits of organic farming diminish at larger scales (Gabriel *et al.*, 2010).

The average effect size and response to agricultural management system depended on taxonomic group, functional group and crop type (Figure 4-7). Plants benefitted the most from an organic approach, while arthropods, birds and microbes also showed a positive effect (Tuck *et al.*, *op cit.*). Among functional groups, pollinators showed the largest effect size, while soil-living decomposers showed little effect. This lack of effect on decomposers is somewhat surprising, given the fact that organic farming has been shown to benefit soil structure and soil conditions (see section 4.5.3), but may be due to stronger influences of soil type and structure on soil biodiversity than the farming system itself, although it was noted that soil organisms were in general understudied.

There were also varying responses among crop types, with large positive effect sizes in cereals and mixed farming and moderate positive effects for all others (Figure 4-7). Of the three measures of land-use intensity (proportion of arable fields; typical field size; number of habitats), only the proportion of arable fields had any significant overall effect, with the difference in diversity between organic and conventional increasing with increasing proportion of arable fields (Tuck *et al.*, *op cit.*). This suggests that the effect of organic farming on biodiversity is greater in intensively managed landscapes, although again, this was found to vary between groups. For example, predators have a greater response to organic farming in intensively managed landscapes while the effect on pollinators does not increase much with land-use intensity (Tuck *et al.*, *op cit*.).

Fuller *et al.* (2005) showed that organic arable fields can support 68-105% more plant species, and 74-153% greater abundance, compared with conventional arable fields. Roschewitz *et al.* (2005) concluded that as organic systems are characterised by diverse seed banks, organic fields could be viewed as self-sufficient ecosystems for plants, therefore not relying on immigration from surrounding habitats to maintain species pools.

Positive effects of organic farming on plant diversity have been linked to organic management practices including prohibition of herbicide or mineral fertiliser inputs, sympathetic management of non-cropped areas, and more mixed farms (Hole *et al.*, 2005).

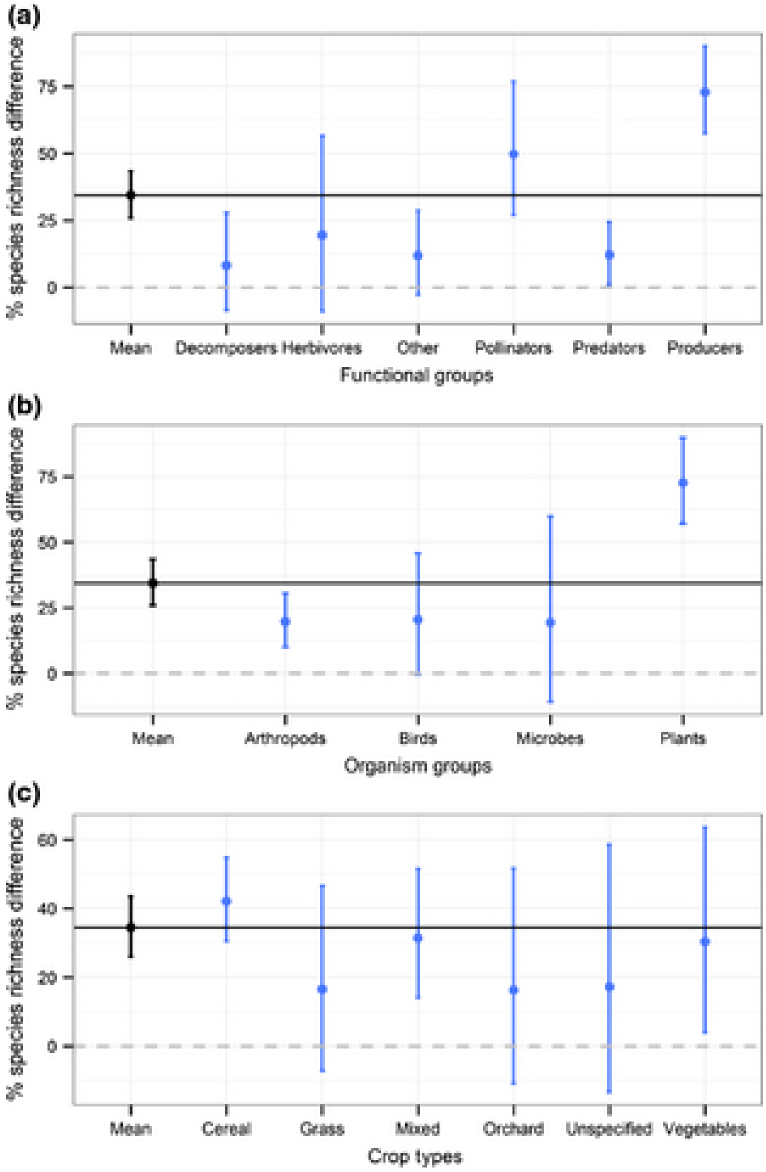


Figure 4-7: The difference in species richness (%) on organic farms, relative to conventional, classified by (a) functional group, (b) organism group and (c) crop types (the grand mean is shown in black, accompanied by the black line; Source: Tuck *et al.*, 2014)

An observation by Ulber *et al.* (2009) was that the increased plant diversity on organic farms arose from multiple aspects of the system, such as longer crop rotations and the absence of herbicides and synthetic fertilisers. This was emphasised by the observation that, under non-organic conditions, a change of only a single factor, in this case the introduction of crop rotation, did not affect plant diversity.

Organic farms tend to have more favourable habitats such as hedgerows, grass margins, grassy ditches, small fields etc. than conventional farms. Norton *et al.* (2009) studying farms in England that had some arable crops found that the organic farms were located in more diverse landscape types, had smaller field sizes, higher, wider and less gappy hedgerows subjected to less frequent cutting, use rotations that include grass, and are more likely to be mixed. Even within diverse landscapes, organic systems had greater field and farm complexity than non-organic systems. This has prompted considerable discussion about whether benefits are derived from the farming system or from the habitat that is independent of the farming system. Some researchers (e.g. Chamberlain *et al.*, 2010) argue that the benefits of organic farming – in this case for farmland bird populations - come "*primarily through greater habitat heterogeneity*" and not from organic farming practice, but as discussed below, even in diverse landscapes, organic farms are more complex.

Several studies covering the range of taxa found that the biodiversity benefits of organic systems are of particular value in simple agricultural landscapes where organic farms are both spatially and temporally more diverse than their conventional counterparts (e.g. Batary *et al.*, 2010; Boutin *et al.*, 2008; Clough *et al.*, 2007b). However, Norton *et al.* (2009) found that even within diverse landscapes, organic systems had greater field and farm complexity than non-organic systems. Some studies have also shown that organic farms can influence biodiversity in the surrounding landscapes with higher diversity recorded on conventional farms in organic 'hotspots' (e.g. Gabriel *et al.*, 2010; Hodgson *et al.*, 2010; Rundlöf *et al.*, 2008) i.e. species are ranging across neighbouring conventional farms.

Hawes *et al.* (2010), in research comparing the diversity and abundance of the within-field seedbank and emerged weed flora on conventional, integrated and organic farms across the arable east of Scotland, found significant responses to management intensity, ranging from high agrochemical input use and winter cropping to no inorganic inputs, spring cropping and mixed farming practices. Within fields, species richness was greatest in organic farms, where there were more weeds. However, at a regional and landscape scale, species richness was greater in integrated and conventional farms, due in part to a greater range of crop types and cropping practices between fields, particularly on the integrated farms.

Organic farming has been found in general to have a positive impact on bird biodiversity, but with variations in responses reflecting species-specific responses. To some extent this may be due to the scale of physical weed control on organic farms (e.g. Geiger *et al.*, 2010) but could also be partly due to the size and mobility of birds together with specialisation of habitats. Gabriel *et al.* (2010) recorded higher overall diversity on conventional farms (particularly of farmland specialists), despite greater food resources (arthropod abundance, weed seeds, and a higher proportion of winter stubble) in the paired organic farms. However, generalist species and members of the crow family were found in higher densities on organic farms. They concluded that landscape characteristics, such as the proportion of arable land and semi-natural grassland, and field margin and hedge lengths, rather than farm management appeared to be the important drivers of bird abundance, although these may in part be a function of management system.

In a study of field-breeding birds, Kragten and de Snoo (2008) found higher abundances of skylark on organic farms, reflecting this species preference for spring cereals which are generally perceived to be more widespread in UK organic systems. With a focus on upland farms in England and Wales, Watson *et al.* (2006) found that in winter, there were significantly higher total densities of birds, and in particular insectivores and Farmland Bird Indicator species, on organic farms. In the non-cropped environment, the longer and more varied hedges that tend to characterise organic farms do have some advantages for a range of bird species relative to non-organic farms, especially in simple landscapes (Batary *et al.*, 2010). Invertebrate-feeding species particularly benefit from the greater habitat diversity found in organic systems, which enhance foraging resources (Smith *et al.*, 2010). In Scottish research, McCracken and Tallowin (2004) highlighted the importance of mixtures of grasses and broad-leaved plants with a range of vegetation heights and structures to provide plant and invertebrate food sources for farmland birds. The diverse ley mixtures including herbs and legumes favoured by some organic producers may support this process, although cutting and grazing management could counteract any benefits. Bird populations may also be influenced by the use of untreated seed on organic farms.

Pollinating insects, such as butterflies and bees, particularly seem to benefit from organic practices (Feber *et al.*, 2007; Rundlöf *et al.*, 2008; Rundlöf and Smith, 2006; Hodgson *et al.*, 2010; Holzschuh *et al.,* 2007; Gabriel *et al.*, 2010; Clough *et al.*, 2007a), probably reflecting the greater floral resource base available both within the cropped area and semi-natural habitats (see previous section on plants).

Predatory taxa including spiders, wasps and ground beetles also respond positively to organic farming (Schmidt *et al.*, 2005; Holzschuh *et al.*, 2007; Diekötter *et al.*, 2010) which has been attributed to greater structural diversity within habitats, increased habitat connectivity and the availability of overwintering habitat and alternative feeding resources in semi-natural habitats.

A minority of studies have recorded no significant differences, or a negative response to organic systems, reflecting taxon-specific variation. Ground and rove beetles, pests, and parasitoids have been recorded in lower densities on organic farms in some studies (Fuller *et al.*, 2005; Clough *et al.*, 2007a; Bengtsson *et al.*, 2005). Species of ground and rove beetles vary widely in their habitat preferences (Luff, 1996) and some species may prefer conditions found on conventional farms.

Macfadyen *et al.* (2009b) found that herbivores in organic fields were attacked by more parasitoid species, while Crowder *et al.* (2010) found that pest control was due to greater evenness of natural enemy populations, independent of species richness. Success is, however, variable (Macfadyen *et al.*, 2009b; Roschewitz *et al.*, 2005; Macfadyen *et al.*, 2009a), perhaps because of unrecognised interactions in multitrophic feeding systems and because of the complex interaction between dynamics of their hosts and responses to local and landscape factors (Holzschuh *et al.*, 2007).

4.4.4 Agroforestry

Agroforestry systems, by their nature, are more diverse than monocultures of crops and livestock. This increase in 'planned' biodiversity (the components chosen by the farmer) increases levels of 'associated' biodiversity (the wild plants and animals also occurring on the farmland).

For farmland biodiversity, research has found that scattered trees within agricultural landscapes act as 'keystone species' that facilitate the movement of wildlife through a landscape that may otherwise be too hostile (Manning *et al.*, 2009). By integrating trees within the agricultural matrix, agroforestry can provide corridors that allow movement of species through landscapes. This role will increase in importance under predicted climate change scenarios by allowing species to adapt their distributions in response to the shifting climate.

A study from the Americas suggests that the impact of agroforestry on biodiversity may extend beyond the landscape-scale. Perfecto *et al.* (2009) consider the correlation between decreasing populations of songbirds in the eastern USA and the elimination of shade trees from coffee agroforests in Latin American countries. Those species in decline were migratory species that overwintered in the southern countries, and were found in the forest-like habitats of traditional coffee farms with a diversity of shade tree species.

The value of agroforestry for UK biodiversity has been assessed in a number of studies on trial sites in the late 1990s (Burgess, 1999; McAdam and McEvoy, 2008; McAdam *et al.*, 2007). Within poplar silvopastoral systems, botanical composition of the understorey changed as the trees matured, with swards dominated by *Agrostis capillaris, Holcus lanatus* and *Poa annua* under the tree canopy while *Lolium perenne, Poa trivialis, Trifolium repens* and *Cirsium arvense* were more common in open pasture (Crowe and McAdam, 1993). Higher abundance and species richness of invertebrates were recorded in silvopastoral systems compared to open grassland in Northern Ireland and Scotland (Cuthbertson and McAdam, 1996; Dennis *et al.*, 1996).

On a silvoarable trial site in Silsoe, Bedfordshire, common arable weeds including barren brome (*Bromus sterilis*), blackgrass (*Alopecurus myosuroides*) and common couch (*Elymus repens*) colonised the area under the trees (Burgess, 1999). Although these species have value as a resource for farmland biodiversity, they can potentially be a nuisance for farmers if they act as a major seed source for reinfestation of the field. With careful and planned management, including sowing with more desirable species and periodic mowing, weed species can be managed in the area under the trees so as not to present a problem for farm production. Peng *et al.* (1993) recorded higher abundances and species richness of airborne arthropods in a silvoarable system compared to an arable control in northern England, probably in response to a greater diversity of plants along the tree rows. Some taxa have more species-specific responses. Phillips *et al.* (1994) found that some species of carabid beetles were more common in the agroforestry system, while others were more common in the arable control. This is likely to reflect the narrow habitat requirements of many carabids. Some species prefer open habitats found in arable fields, others prefer damp, shaded conditions associated with tree cover.

Vegetated understoreys within the tree rows were shown to have higher abundances of spiders than bare understoreys in a silvoarable system in Yorkshire (Burgess *et al.*, 2003), reflecting the association of this taxon with habitats with greater structural diversity. The number of bank voles (*Clethrionomys glareolus*), wood mice (*Apodemus sylvaticus*), field voles (*Microtus agrestis*) and common shrews (*Sorex araneus*) were higher in the silvoarable system than in the arable and forest control areas, possibly reflecting an edge effect (Klaa *et al.*, 2005; Wright, 1994). Even at an early stage of development, the silvopastoral systems were shown also to have a positive impact on birds, and attracted both woodland and grassland species, thus creating a unique assemblage of species (McAdam *et al.*, 2007). More recently, preliminary results from a PhD research project investigating ecosystem services in six organic agroforestry systems in England indicated significantly higher abundance and species diversity of butterflies in agroforestry sites compared to the monocropping controls (Varah *et al.*, 2013).

Using pollinator abundance and species diversity as a proxy for pollination services, Varah *et al.* (*op cit.*) recorded significantly higher pollinator abundance in silvoarable systems. This was attributed to the development of understorey vegetation within the rows of trees. The silvoarable understorey of grasses and forbs remains largely undisturbed in these organic agroforestry systems, allowing greater structural diversity and flowering plants to reach maturity, thus providing nesting habitat and foraging resources for many pollinator species (Varah *et al.*, 2013).

4.4.5 Factors affecting the biodiversity and related ecosystem service impacts of agroecological approaches

The factors influencing the biodiversity value of agroecological approaches are primarily derived from: increasing spatial and temporal diversity of the farming system; provision of permanent habitats and areas with lower disturbance; and specific management practices including the reduced or non-use of pesticides. Key factors are summarised in Table 4-8.

Spatial and temporal diversity within the farm can be enhanced:

- within species (e.g. composite cross populations, variety mixtures);
- between species (e.g. cereal/legume mixtures, diverse fertility-building leys, polycultures);
- at the system level (e.g. crop rotations, mixed farming, agroforestry) and
- through the management of ecological focus areas or non-cropped habitat (hedges, ditches, farm woodland, beetle banks, field margins)

Increasing the planned agricultural diversity within the farm leads to higher levels of associated biodiversity (i.e. wild species existing on the farmland). These species could be beneficial (e.g. pollinators, natural enemies), detrimental (pests) or neutral (e.g. some bird species). In turn, these contribute to the ecosystem services that underpin agroecological farming. The greater diversity of components within the farming system, and of non-crop habitats (such as beetle banks, pollen and nectar mixes, diverse legume mixtures and wildlife seed mixes), is not only with the perspective of supporting wildlife *per se*, but directly contributes to supporting the farming system, including soil fertility building, crop protection and animal health maintenance.

Table 4-8: Factors affecting biodiversity and related ecosystem service impacts of agroecological compared with intensive conventional systems

Output parameter	*Integrated*	*Organic*	*Agroforestry*	*Key factors*
Soil micro organisms	+	++	++	Cultivations, supply of organic matter as energy source for soil ecosystem, distribution of nutrients within soil profile
Invertebrates	++	++	+++	Cultivations, supply of organic matter, storage and application of slurries/ manures, soil pH, provision of undisturbed field features (e.g. beetle banks), restricted use of pesticides
Plants	+	++	+++	Specific habitats (beetle banks, field margins), non-use of herbicides, crop species and variety choices, rotations and polycultures
Pollinators	+	++	++	Provision of food sources throughout season, either as pollen/nectar mixes, flowering strips, or diverse legume mixtures; restricted use of pesticides
Mammals	+	+	++	Availability of non-cropped habitats, corridors e.g. hedges, farm woodland, tree lines, as well as permanent grassland
Farmland birds	+	+	+/-	Alternation of winter and spring crops, provision of nesting sites for ground-nesting birds, restricted use of pesticides, feed availability in hungry gap

- = less than conventional, 0 = similar to conventional, + = higher than conventional

Source: Own assessment based on literature presented in this section.

As part of this, the provision of permanent, semi-natural habitats (e.g. hedgerows, farm woodlands, ponds and ditches) and cropped areas with lower disturbance (e.g. permanent pastures, medium and long term fertility-building leys, tree rows in agroforestry, beetle banks, field margins, floral strips and headlands) will provide shelter, nesting and feeding resources for farmland biodiversity, as well as providing connectivity to support wider biodiversity within the landscape.

Specific agricultural management practices will impact biodiversity both directly (e.g. physical damage through cultivation) as well indirectly (e.g. by modifying the environment or reducing/increasing feed resources). Beneficial practices include:

- The avoidance of agrochemical inputs; both pesticides and soluble fertilisers
- The use of legumes
- Restricted use of slurry and manure applications
- Reduced or zero tillage
- Promotion of high soil organic matter

Some of these practices are common to almost all agroecological approaches, but others, such as the non-use of herbicides and most pesticides, are restricted to specific approaches, such as organic and biodynamic farming and permaculture. This leads to within field biodiversity benefits, as well as on field margins, where the primary benefits of integrated crop management would be expected. However, the avoidance of herbicides also makes the adoption of other practices such as reduced/zero tillage more difficult, although the biodiversity losses (e.g. earthworms) at this stage may be compensated by the fertility-building phase of the organic rotation.

Most studies consider the impact of agricultural practices on individual ecosystem services, and few consider trade-offs or synergies between services. One recent study has investigated the effect of one agroecological practice, polycultures, on two ecosystem services, biocontrol of herbivorous pests and yield of a focal crop, to identify whether this practice promotes a trade-off or win-win relationship between these two services (Iverson *et al.*, 2014). Using a meta-analysis approach, the authors found a win-win relationship between yields and biocontrol in polyculture systems that minimised intraspecific competition via substitutive planting. Even under high cropping densities (through additive planting), there were beneficial effects on biocontrol with no difference in crop yields, as long as legumes were used as the secondary crop.

While the evidence for the biodiversity benefits of agroecological approaches seems clear, there has been intense debate as to whether the lower yields from some of these systems compared to conventional, high input systems lead to a trade-off between food production and nature conservation (Gabriel *et al.*, 2009; Green *et al.*, 2005; Badgley *et al.*, 2007; Bengtsson *et al.*, 2005; Dobermann, 2012; Gabriel *et al.*, 2013; Hole *et al.*, 2005b; Mondelaers *et al.*, 2009; Reganold, 2012; Tuomisto *et al.*, 2012; Winqvist *et al.*, 2012).

Green *et al.* (2005) identifies two alternative management strategies for conservation of biodiversity; 'land-sharing' or 'land-sparing'. In a 'land-sharing' strategy, production techniques are used to maintain some biodiversity throughout agricultural land while in a 'land-sparing' approach, some land is set aside for conservation while other land is used intensively to produce agricultural goods. Agroecological approaches, such as organic farming, provides shared benefits to both humans and wildlife and fall under the 'land-sharing' approach, while conventional farming maximises yields, and thus spares wild land elsewhere in a land-sparing approach. The 'best' strategy depends on the balance between a species population size and farming intensity, so that if a slight decrease in farming intensity (and subsequent drop in productivity) causes a considerable increase in the population size of a wild species, land-sharing is the optimal strategy. Conversely, if a large decrease in intensity resulted in minimal population gains, land-sparing is the best option.

While the debate recognises that some species and habitats will not survive any agricultural interventions and need to be protected separately from agriculture (e.g. Lawton, 2010), it fails to acknowledge the potential role of biodiversity in maintaining productivity through the ecosystem services it supports, such as pollination, pest control and nutrient cycling on land already used for agriculture. As Tscharntke *et al.* (2012) put it:

"*A major argument for wildlife friendly farming and agroecological intensification is that crucial ecosystem services are provided by 'planned' and 'associated' biodiversity, whereas the land sparing concept implies that biodiversity in agroecosystems is functionally negligible.*"

As recent work regarding the impact of intensive farming practices on key pollinators (Vanbergen and Initiative, 2013) has demonstrated, it is not simply a case of choosing between biodiversity and productivity – both of these things are interdependent. Other species that are characteristic of extensively managed farmland would clearly be threatened by agricultural intensification and therefore require a land-sharing approach (Chamberlain *et al.*, 2000); such species (e.g. farmland birds) are often integral parts of our cultural landscape and so feature high on the agenda for policy makers.

In recent years, the argument has moved away from a straightforward distinction between land sharing and land sparing approaches (Fischer *et al.*, 2013). For example, the question of whether an approach is defined as sharing or sparing depends on an interpretation of scale. Landscapes considered by some authors to be examples of 'land-sparing' are seen by others as examples of 'land sharing'. These considerations may be influenced to some extent by the taxa in question, and at what scale they respond to the landscape change (e.g. birds will respond very differently by comparison with soil micro-arthropods). Additionally, land-use within any given region is affected by more distant drivers so that the trade-off between agriculture and biodiversity may actually occur in separate locations (Fischer *et al.*, 2013). As a result, effects of implementing either land-sparing or land-sharing in a particular region may be displaced to another region.

Herzog and Schüepp (2013) argue for the need to differentiate between productive and marginal farmland and propose that land-sparing is appropriate for neither type. On highly productive farmland, semi-natural habitats are required to support ecosystem services relevant for agriculture, to safeguard threatened farmland species that are important to society, and to allow migration of non-farmland species through the agricultural matrix. However, there will be a trade-off between production and biodiversity and it is difficult to define the minimum level of biodiversity required in such contexts. This could also include access and recreation services on farmland in the UK, as the population is dense and people live in all areas. On more marginal land, high-nature value farming is a traditional farming practice which produces high quality, culturally and geographically distinct agricultural products, conserves specialised species and has high cultural and recreational value (Herzog and Schüepp, *op cit.*).

4.5 Soil and water resources

4.5.1 Assessing soil and water resource use impacts

In this section, we focus on the more physical attributes of soil and water resources, with the carbon sequestration and biological aspects addressed in other sections above. With respect to soils, key issues are the impacts of agroecological practices compared with other approaches to sustainable intensification on soil nutrient status (fertility), structure, infiltration, compaction and erosion. Soil biota such as earthworms and microbes will impact on aggregate stability, erosion and water quality (these issues are considered in more detail in sections 3.2 and 4.4).

With respect to water, key issues include eutrophication and food web modifications, pesticide pollution, increased sediment load from soil erosion, changes to hydrological cycles via changes in evapo-transpiration rates and run-off, modification of river flow and irrigation impacts, effects of exotic and non-native species, and physical modification of the habitat through cannelisation, drainage and embankment (Moss, 2008).

4.5.2 Integrated crop/farm management

Reduced tillage is an important component of integrated crop management and conservation agriculture. Over the past decade, the UK Soil Management Initiative has gathered a substantial body of evidence regarding the impacts of reduced tillage cultivation systems. Much of this information is published in the SMI soil management guides (SMI, 2005; Vaderstad and SMI, 2004; 2006), along with numerous papers and articles in the scientific literature and farming magazines.

Montgomery (2007) concluded after a review of studies in various countries, that run-off and erosion rates of agriculturally managed soils (applying conventional ploughing and soil management strategies) are 10 to 100 times higher than the rates observed under native vegetation or general long-term geological erosion. Whereas the latter is normally balanced out by natural soil build-up and production, erosion levels in plough-based agricultural systems have been found to be unsustainable. However, they conclude that reduced tillage and no-till farming approaches can provide useful, more sustainable alternatives, with lower erosion rates approaching those under native vegetation or soil production.

Managing crop residues is key to total resource management systems. The previous year's residues benefit the current crop because when they decompose they add both nutrients and organic matter to the soil. In time, incorporation of crop residues in the topsoil layers will lead to improvements in soil structure and a better soil tilth. Topsoil profiles are the most biologically active, containing many micro-organisms and earthworms, which lead to improvements in nutrient recycling and soil porosity. Conservation tillage systems start each year with the production and distribution of residue from the previous year's crop.

Farmers report that the increase in crop residues at the soil surface creates, over time, a higher level of soil organic matter in this critical zone, making working the soil easier particularly in dry conditions. Two studies on zero-tilled land have shown significant increases in soil organic matter. Longhurst (2010) showed 20 times more earthworms in three fields of Denchworth series clay compared to ploughed comparisons nearby, giving rise to greater water infiltration; and Carel (2012) recorded organic matter levels of over 30% in the top 20 cm of a zero-tilled silty loam soil compared to less than 5% in the ploughed comparisons. Alton (2006), using soil taken from the site of the *Soil and Water Protection Project* (SOWAP) (Jones *et al.,* 2006), which consisted of a series of farm-scale erosion plots comparing plough based tillage with minimum tillage, subjected them to rainfall simulation in laboratory conditions. The minimum-tilled plots showed reduced erosion and analysis indicated this was due to increased biological function in the soil.

From 2003-2007, the SOWAP project measured the impact of three cultivation regimes on two farms in the UK, one farm in Belgium and one farm in Hungary. The regimes consisted of plough based, non-inversion and "farmer preference" treatments. The results across the four sites were variable, with spring sown crops showing higher levels of erosion than autumn drilled and the plough based systems also tending to give greater levels of erosion. However non-inversion cultivations tended to produce run-off earlier in the autumn than the plough treatments although the sediment load always tended to be lower. Measurements with a soil penetrometer showed that min-till soils had greater bulk density in the top 10cm compared to ploughing, but this is unsurprising in the conversion period from conventional tillage. Worm densities and biomass grew significantly under lower intensity cultivation over the life of the project (Rothwell, 2007) reflecting the observations of other studies (Jordan *et al.,* 2000a). These are likely to increase the distribution of organic matter in the soil as well as providing conduits for water absorption.

4.5.3 Organic farming

<u>Soil conservation</u>

Good soil structure and physical condition is highly dependent on soil organic matter content (Arden-Clarke and Hodges, 1988; Gattinger *et al.*, 2012; see also section *4.3* above). Soils in organic farming systems receive regular inputs of organic material and various studies have shown that organic farms often have a better soil structure than conventionally managed farms (Shepherd *et al.*, 2002; Pulleman *et al.*, 2003; Gattinger *et al.*, 2012). Research in the United States and New Zealand (Reganold *et al.*, 1987; 1993) has also identified soil organic carbon, soil structure and soil erosion benefits on land under organic or biodynamic management compared with conventional.

According to Shepherd *et al.* (2002), the higher soil organic matter content on organic and biodynamic farms can be mainly attributed to regular organic additions, as well as leys in the rotation. However, in farming systems relying on inorganic fertiliser inputs, in particular nitrogen, organic matter levels can be difficult to maintain, due to breakdown by soil microbes in the presence of good nutrient supplies, resulting in poor soil structure, reduced crop root penetration and water retention capacity. Organic fertilisation strategies tend to sustain and improve soil structure, are less disruptive to the soil chemical environment, increase biodiversity of the soil biota and suppression of soil-borne pathogens (Arden-Clarke and Hodges, 1988). However, although mineral sources of P, K and other macro and micro-nutrients are permitted, they may not be used in sufficient quantities to compensate for nutrients exported from the holding, leading to a risk of mining nutrients over time (see also section 3.2.1).

Mäder *et al.* (2002) defined soil fertility as the provision of essential nutrients to crop plants, the support of a diverse soil biota as well as showing a typical soil structure and decomposition rates. In a long-term systems comparison trial in Switzerland (DOK), the team found that lower energy costs are required to transform carbon from organic material in organically managed soils compared to conventionally managed soils; concluding that more diverse soil microbial communities are building higher levels of microbial biomass (Mäder *et al.*, *op cit.*).

Water quality

Due to the strict limitation of chemically synthesised inputs in plant production, organic farming significantly helps reduce residues of plant protection products and chemical fertilisers in water, thus improving water quality (Mahé and Portet, 2012). Rotations including legumes and green manures, the use of farmyard manure as fertiliser and the limitation of stocking densities and total amount of livestock manure reduce the input and availability of rapidly soluble nitrogen, and therefore reduce leaching of nitrates. Several studies show that nitrate leaching can be reduced by 40–64 % through organic farming (e.g. Edwards *et al.*, 1990; Younie and Watson, 1992; Eltun, 1995; Condron *et al.*, 2000; Goulding, 2000; Haas *et al.*, 2001; Kirchmann and Bergström, 2001; Mäder *et al.*, 2002; Stopes *et al.*, 2002; Auerswald *et al.*, 2003; Pacini *et al.*, 2003; Shepherd *et al.*, 2003; Osterburg and Runge, 2007).

Based on a statistical comparison of 12 studies, Mondelaers *et al.* (2009) concluded that the nitrate leaching rate is on average 9 kg ha^{-1} in organic production versus 21 kg ha^{-1} in conventional agriculture. Important differences are noted among the studies due to differences in soil, regions, fertilisation practices and measurement. In contrast to the results mentioned above, in some comparative crop rotation experiments nitrate leaching has been reported at the same levels in organic and conventional rotations (Korsaeth and Eltun, 2000), especially if calculated per kilogram of harvest (Mondelaers *et al.*, 2009).

Looking at the impact per kg output, Nemecek *et al.* (2005) found higher eutrophication impacts per output for some organic crops compared to conventional. In some places, these higher nutrient loads on arable land are attributed to the greater use of organic fertilisers in the organic system, because the life cycle assessments used by Nemecek et al. (2005) assume relatively high fertilisation rates for organic farms (to compensate for nutrient offtakes). Taking Nemecek *et al.*’s (2005) data and projecting them at sector level, using statistical data and an economic model, Schader (2009) found on average 35 % lower eutrophication rates on organic farms per hectare. The following facts underline the lower eutrophication potential of organic farming found in literature (Schader et al., 2012):

- Organic farming systems have lower nutrient application levels, which reduce the absolute quantity of nutrient loads that can be emitted from the system due to the prohibition of synthetic nitrogen fertilisers, lower stocking rates and restrictions on the use of manure;
- The quantity of directly available nitrogen is much lower in organically managed soils;
- Because nutrients cannot be imported easily into the systems, the opportunity cost of nitrogen losses is higher for organic farms than for conventional farms (Stolze et al., 2000). This implies a need for more efficient nutrient management in organic systems, although this does not eliminate losses. In addition, nitrate leaching can be high at the point of transition from the fertility building phase of the rotation to the cropping phase.

In animal husbandry, outdoor production of pigs and poultry (not specifically organic but with access to pasture) increases the risk of nitrate losses if excrements are concentrated in certain areas and vegetation cover is allowed to deteriorate (Eriksen *et al.*, 2002; 2006; Degré *et al.*, 2007; Salomon *et al.*, 2007; Halberg *et al.*, 2010). In organic systems, livestock stocking rates are restricted by the EU organic regulations to the equivalent of 170 kg N ha^{-1}, the same limits as in the Nitrates Directive. However, the report of the European Commission's *Expert Group for Technical Advice on Organic Production* (EGTOP) on poultry pointed out that the minimum outside area for laying hens of 4 m^2 can sometimes lead to a pressure of nitrogen that exceeds 170 kg ha^{-1} $year^{-1}$ (EGTOP, 2012). For herbivores, the maximum stocking density (related to the limit of 170 kg N/ha) is implemented at the farm level, but higher stocking rates may occur on specific fields.

There could be a positive impact of organic production practices in relation to water use, partly related to production rules. For example, Stanhill (1990) and Lotter (2003) found that organic crops show higher ability to cope with drought than conventional ones, mainly because organic farming practices commonly increase and stabilise soil organic matter. More recently, a French study comparing 151 organic holdings to 281 conventional ones (Caplat, 2006) found that only 8 % of the organic areas were irrigated, whereas 33 % of conventional holdings used irrigation.

Hathaway-Jenkins *et al.* (2011) and Hathaway-Jenkins (2011) have shown that soil water infiltration rates are greater under organically managed grassland than conventional. Organic management improved soil quality for maximum water holding capacity, aggregate stability, shear strength and infiltration rate. However, there was no significant difference due to organic management for other soil quality parameters such as bulk density, field capacity, plastic limit, total porosity, pH, total C:N ratio or workability. Infiltration rates were higher or equal to conventional arable land and this could be related to the significant improvement in maximum water holding capacity for organically managed soils. This has implications for flood prevention, as organically managed land has an increased capacity to store water.

4.5.4 Agroforestry

Research has demonstrated that agroforestry vegetation buffers can reduce pollution from crop fields and grazed pastures (Udawatta *et al.*, 2002; Lee and Jose, 2003; Anderson *et al.*, 2009; Dougherty *et al.*, 2009; Udawatta *et al.*, 2010). Riparian buffers in particular, can reduce non-point source water pollution from agricultural land by reducing surface runoff from fields; filtering surface and groundwater runoff and stream water, and reducing bank erosion (Dosskey, 2001).

The 'safety net hypothesis' is based on the belief that the deeper-rooting tree component of an agroforestry system will be able to intercept nutrients leached out of the crop rooting zone, thus reducing pollution and, by recycling nutrients as leaf litter and root decomposition, increasing nutrient use efficiencies (Jose *et al.*, 2004). Greater permanence of tree roots means that nutrients are captured before a field crop has been planted and following harvest, when leaching may be greater from bare soil.

Buffer strips can significantly decrease pollution run-off, with reductions of 70-90% reported for suspended solids, 60-98% for phosphorus and 70-95% for nitrogen (Borin *et al.*, 2009). A study in central Iowa, US, found that a switch-grass/woody buffer removed 97% of the sediment, 94% of the total N, 85% of the nitrate-N, 91% of the total P and 80% of the phosphate P in the runoff (Lee *et al.*, 2003). Agroforestry systems also have the potential to mitigate movement of harmful bacteria such as *Escherichia coli* into water sources (Dougherty *et al.*, 2009) and reduce the transport of veterinary antibiotics from manure-treated agroecosystems to surface water resources (Chu *et al.*, 2010). Agroforestry has been used to address issues of soil salinisation in Australia where a study recorded a lowering of the saline groundwater table by two metres over a seven-year period under a Eucalyptus-pasture system, relative to nearby pasture-only sites (Bari and Schofield, 1991).

During drought periods, tree roots access deeper soil horizons for water, reduce evapotranspiration from the understorey vegetation and provide shade for crops and livestock. Easterling *et al.* (1997) used a crop modelling approach to look at the effect of climate change on shelterbelt function and found that under several climate change scenarios, windbreaks could help maintain crop production, with sheltered crops performing better than unsheltered crops.

During flooding events, where trees are present as part of agroforestry systems, the tree roots access deeper soil horizons and a larger area than surface crops. When land is flooded the trees work like 'pumps', removing water from the upper soil layer quicker than from land cropped with monocultures. Research at INRA, France has demonstrated that access to land for agricultural purposes after flooding events can be 7-14 days sooner under agroforestry than for land cropped as a monoculture (Dupraz *et al.,* pers. comm.).

Research to investigate the impact of land management changes on soil hydrology and flood risk was carried out at the Pontbren experimental catchments in Wales between 2004 and 2012 (Jackson *et al.*, 2008; Woodland Trust, 2013). Small-scale manipulation plots were used to monitor the hydrological effects of de-stocking and native broadleaf tree planting under controlled conditions. Planting native broadleaved trees significantly improved soil infiltration rates five years after treatment application, with infiltration rates in the tree plots 13 times and 67 times greater than in the ungrazed and grazed plots respectively. This increase in infiltration was attributed to changes in the soil macropore structure and was associated with a reduction in soil bulk density in the upper soil horizons. Associated with increases in soil infiltration were reductions in surface runoff. Land management was also shown to affect stream flow responses with shorter residence times (i.e. flashier stream flow response and increased flood peaks) associated with catchments dominated by improved grassland land use. Using the data from Pontbren, a multidimensional physically-based model has shown how careful placement of small strips of trees within a hillslope can reduce magnitudes of flood peaks by 40% at the field scale (Jackson *et al.*, 2008).

4.5.5 Factors affecting the soil and water resource impacts of agroecological approaches

Table 4-9 summarises the main factors affecting soil conservation and water protection. With respect to soils, the main focus in this section has been on the physical components of soils, with issues relating to soil organic carbon and biological activity covered in previous sections. While the underlying geological basis for soils is independent of the management system applied, the availability, removal and replenishment of nutrients can be influenced by agroecological practices, including the use of organic soil amendments and specific plant species to stimulate the release of nutrients by soil organisms and plant root exudates. While the use of livestock manures may be more prevalent in organic systems, green manures and cover crops are now widely used in both integrated crop management and organic farming, while the deeper rooting systems and return of leaf litter in agroforestry actively contribute to nutrient cycling.

Soil erosion and soil compaction are influenced by tillage practices as well as crop and grazing management, practices that are not unique to specific agroecological systems. To the extent that ground cover is maintained through the use of cover crops and undersowing, soil erosion can be reduced, and agroforestry treelines on contours of sensitive slopes can provide significant protection against erosion. Water stable soil aggregates are also important in maintaining soil structure and reducing soil erosion, with increased earthworm and other soil biological activity making a significant contribution. The use of reduced tillage integrated systems and the use of fertility building leys in organic systems are relevant in this context.

Table 4-9: Factors affecting the soil and water resource impacts of agroecological approaches compared with intensive conventional systems

Output parameter	***Integrated***	***Organic***	***Agroforestry***	***Key factors***
Reduction of soil erosion	+	++	+++	Maintenance of organic matter, earthworm and other soil biological activity, ground cover, tree lines
Reduction of soil compaction	++	+	+++	Reduced tillage, appropriate machinery
Soil fertility improvement	+	++/-	++	Nitrogen and carbon fixation; manures, organic matter and leaf litter return; possibly mining of nutrients in organic systems
Improved water quality	0/+	++	++/-	Avoidance of agrochemicals, lower nutrient applications, reduced erosion; agroforestry may be -ve if herbicide used to maintain vegetation-free understorey
Flood mitigation	0	++	++	Improved soil structure, increased water infiltration, earthworm activity
Improved drought tolerance	0	+	++	Improved soil water holding abilities

- = less than conventional, 0 = similar to conventional, + = higher than conventional

Source: Own assessment based on literature presented in this section.

The different levels of nutrient and agrichemical use in integrated and organic systems contribute to higher risks of nitrate leaching, eutrophication and pesticide residue contamination of water supplies from integrated than from organic systems, although organic systems also have weak points for nitrate leaching when the fertility building leys are cultivated for the cropping phase of the rotation. However, the benefits of organic management in terms of water protection have led to water companies in several countries promoting organic management in water catchments. Depending on understorey management and the management of crops between the trees, agroforestry systems can also make a significant contribution, except potentially where treelines are kept vegetation free through herbicide use.

There is some evidence that increased earthworm activity and increased proportions of grassland in organic systems can benefit water infiltration rates, with potential to alleviate flooding, and that soil organic matter levels and reduced surface nutrient applications can encourage deeper root penetration, protecting crops from drought conditions. Similar impacts can be found in agroforestry systems, with the added benefit of treelines and microclimate effects. It is not clear from the evidence presented what impacts integrated systems have on these outcomes.

4.6 Profitability

4.6.1 Assessment of profitability

Comparisons of the profitability of agroecological systems with general agriculture need to consider factors that are affected by the differences in management and exclude those that are not, for example, endowment of resources such as land, as well as considering data sources and sampling. This can be done using different methods, such as using existing survey data, specialist surveys or monitor farms or modelling.

Whether or not a farm is organic is determined by their participation in organic certification and such farms are identified in datasets such as FADN and the Farm Business Survey (FBN). For the other two agroecological approaches used in this section, there is no similar identification, so the Farm Business Survey or FADN cannot be used for comparisons.

In the following section, some results on profitability are presented for integrated agriculture, organic farming and agroforestry. The first section is based on monitor farms, the second on farm business survey data and the final one on several modelling studies.

4.6.2 Integrated crop/farm management

Integrated farming seeks to optimise the use of inputs by combining cultural, biological, physical and chemical techniques in a rotational approach to reduce the reliance on a single set of control mechanisms and to improve the ratio of inputs to outputs. In common with the organic approach, calculations on profitability are based upon rotations, rather than individual crops, and consequently integrated farmers tend to take a longer term view of their cropping systems.

Key to the uptake of Integrated Farm Management (IFM) is its robustness in dealing with the challenges of weather and the profitability of the system. The nine Integrated Arable Crop Production Alliance (IACPA) sites combined and published their collective results in 1998 (MAFF, 1998). They reached the following conclusions:

- Use of inputs in IFM was lower, attributed mainly to a reduction in the number of pesticide applications and to lower dose rates of agrochemicals and nitrogen. Average reductions across all sites were: fertiliser 18%, insecticides 40%, herbicides 45% and fungicides 52%;
- Costs of inputs were also reduced due to the reduced usage. Average cost reductions were: fertiliser 18%, growth regulators 74%, insecticides 42%, herbicides 38%, fungicides 41%. The cost reduction percentages are different to the reductions in active ingredients as different products are used in the different systems. The high reduction in the amount of growth regulator applied is a function of the use of strong strawed cultivars and more accurate nitrogen usage in the integrated system.
- On average crop yields were lower, but the reductions were small and less than the normal seasonal variations experienced within crop rotations. Quality was not significantly altered;
- Financial performance was found to be more resilient, particularly at lower prices, due to more efficient use of inputs;
- Overall profitability was maintained, on average, across all sites. Three sites experienced reductions in profitability of 10-20%. This was in part due to the testing of new rotations and different crop establishment techniques. Five sites experienced increases in profitability of between 4 and 14%, with one site showing no difference.

More detailed analysis, including individual site results can be found in Jordan and Hutcheon (1994), Keatinge *et al.* (1999), Jordan *et al.* (2000) and Cook *et al.* (2000).

4.6.3 Organic farming

Since the mid 1990s, data on organic farm financial performance in England and Wales has been collected as part of the Farm Business Survey (FBS) and analysed by researchers at Aberystwyth University and the Organic Research Centre, funded until the publication of the 2011/12 report in 2013 by Defra[36]. Data collection on all farms is carried out in the same way following the FBS protocol. The comparison with conventional is based on similar

[36] The annual reports can be downloaded from www.orgprints.org, using the search term 'Organic Farm Incomes'.

conventional farms based on resource endowment (land area, location etc.); i.e. factors that are not influenced directly by organic or conventional management.

A clustering process is carried out to select the comparison farms, because the organic farms in the FBS are selected primarily as part of the process to represent agriculture in general, not organic farming specifically, and organic farms are not distributed across the different farm types in the same proportions as conventional farms. As a result, comparisons of the organic farms against the simple group average for conventional farms of that type could lead to distorted results. (In recent years, the FBS in England has over-sampled organic farms to make sure that there are sufficient numbers for valid samples in each farm type, but the results are then adjusted to compensate when they are grouped with all farms.) A summary of farm business income performance per ha for organic compared with conventional farms of different types is shown in Figure 4-8 (Moakes *et al.*, 2014).

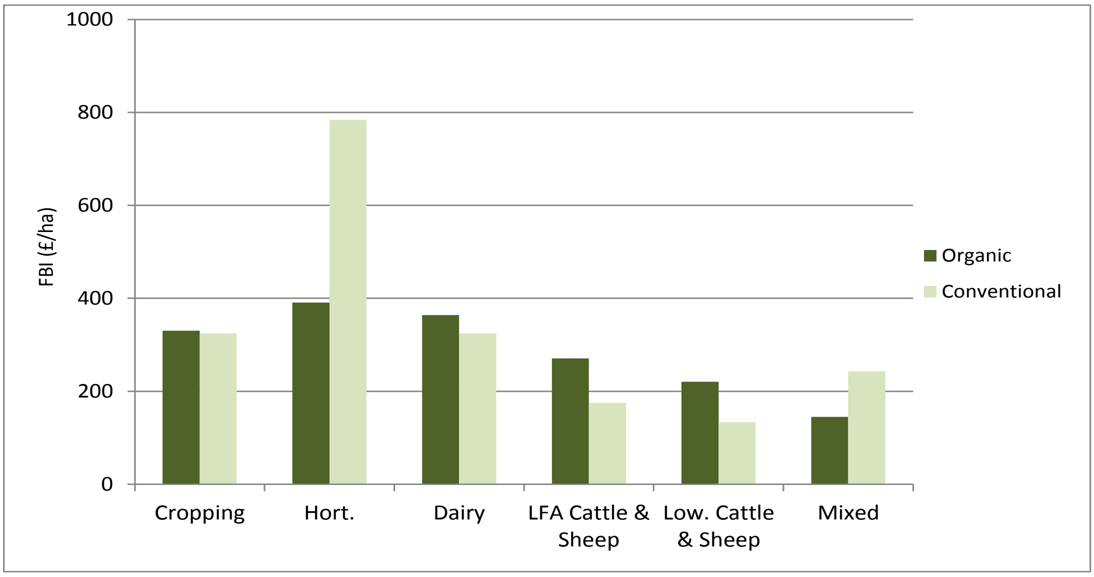


Figure 4-8: Organic and conventional Farm Business Income (£/ha, full samples, 2012/13) (Source: Moakes *et al.*. 2014)

An analysis of longer-term trends from 2006/07-2011/12 (Lampkin *et al.*, 2014) supplemented by the 2012/13 data (Moakes *et al.*, *op cit.*) shows that for most farm types, the profitability of organic farms has held up much better and shown less volatility than might have been expected during the recession, despite its impact on the UK retail market for organic food. The performance of organic farms remained comparable with that of similar conventional farms for most farm types (see Figure 4-9, a-d), though horticulture fared much worse than others relatively. Pigs and poultry farms were not reported due to insufficient sample size.

Although the FBI performance per ha for organic horticultural holdings was significantly worse than conventional (Figure 4-9 b), and highly variable from year to year, this may be a factor of small sample size. Despite this, labour productivity (financial output per full-time labour unit) for both horticultural samples was almost identical over the same period, indicating that business size and intensity may be a more important factor determining employment than management system (see also section 4.2.5).

Since the 1990s, organic farming's financial performance has relied on engagement with consumers, resulting in a price premium for organic produce, as well as agri-environment support for organic conversion and maintenance. While there is potential for reducing input costs, particularly with respect to fertilisers and pesticides, other costs including purchased organic seeds and feeds are often higher. Labour and machinery costs may also be higher, though normally not on a per ha basis. However, when spread over reduced yields, the cost per unit product will be higher.

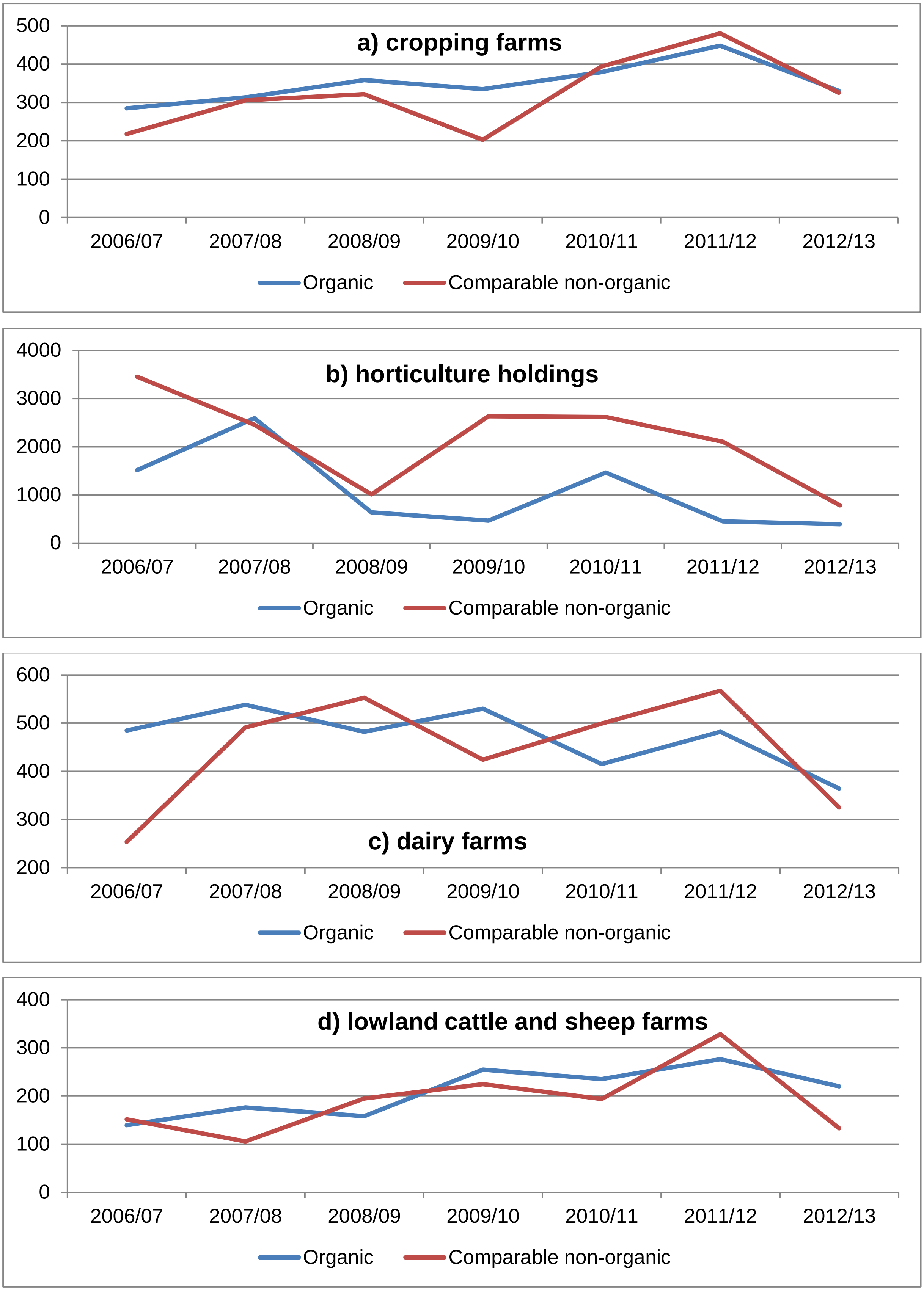


Figure 4-9: Farm Business Income (£/ha) trends for organic a) cropping, b) horticulture, c) dairy and d) lowland cattle and sheep farms compared with comparable conventional farms, 2006/07 to 2012/13. (Sources: Lampkin *et al.*, 2014; Moakes *et al.*, 2014)

The Achilles heel for many organic cropping and horticultural farms is the lack of income generation from the fertility-building phase, especially where livestock enterprises are not a component of the farming system, although in some cases, there may be opportunities to generate a financial return by using the vegetation produced as biomass for energy (biogas) production. There may also be reduced working capital requirements for crop storage (due to lower outputs) and as a consequence of reduced livestock numbers, which might also result in reduced labour requirements (Lampkin *et al.*, 2014).

4.6.4 Agroforestry

Economic studies of agroforestry systems have shown that financial benefits are a consequence of increasing the diversity and productivity of the systems, influenced by market and price fluctuations of timber, livestock and crops. Where agroforestry is introduced into agriculture-dominated regions, there may be issues with access to or development of suitable markets for the tree projects. The costs of establishment, and time delay before returns from the tree components can be realised, may also act as a barrier to the adoption of agroforestry, at least in the absence of support such as that potentially available under EU Rural Development programmes (Smith *et al.*, 2013b).

A New Zealand study comparing the economics of grazing sheep and beef in open pasture with three silvopastoral systems involving *Pinus radiata*, *Eucalyptus fastigata* or *Acacia melanoxylon*, demonstrated that the silvopastoral systems produced higher long-term returns than the open pasture system (Thorrold *et al.*, 1997, in Benavides *et al.*, 2009). A bioeconomic model (*MAST: Modelled Assessment of Swine and Trees*) of a theoretical integrated domestic pig/woodland edge enterprise in the UK suggested that the financial performance of this agroforestry system could be superior to that of a pasture-based enterprise (Brownlow, 1994; Brownlow *et al.,* 2005). The authors identified key factors influencing the profitability of the system: premium prices for 'forest-reared' pig carcasses; the effect of shelter on feed conversion rates; and the availability of cheaper land rents.

Compared with exclusively forestry land use, agroforestry practices are able to recoup initial costs more quickly due to the income generated from the agricultural component (Rigueiro-Rodríguez *et al.*, 2008). Fernández-Núnez *et al.* (2007, in Rigueiro-Rodríguez *et al.*, 2008) carried out an assessment of initial investments and establishment costs of forestry, agriculture and agroforestry in the Atlantic area of Spain. They found that establishing agroforestry required higher initial investment than the agricultural and forestry systems due to higher initial inputs, but over a 30 year period, profitability per hectare was higher in the agroforestry system than in the exclusively livestock (17%) or forestry (53%) systems. When environmental and ecological benefits were included in the evaluation, the performance of the agroforestry system was even higher.

In silvoarable systems, annual returns from crops produced between tree rows can offset plantation establishment costs. Similarly, providing saplings are protected from livestock damage, integrating chickens into newly established plantings enables farmers to receive income well before any income from tree products is realised (Yates *et al.*, 2007). Valuable timber trees such as black walnut (*Juglans nigra*) were once raised as a retirement crop; farmers would sell a mature timber stand to fund their retirement (Scott and Sullivan, 2007). High value timber trees such as black walnut are not ready for harvest until decades after establishment; integrating crops and/or livestock into the system can produce economic value for at least the first twenty years after establishment.

Modelling of economic returns from a black walnut alley cropping system in Midwestern USA highlighted the importance of system design and management for maximising productivity (Benjamin *et al.*, 2000). Systems with widely-spaced tree rows (12.2m between tree rows) were predicted to be more profitable than both closely-spaced (8.5m between tree rows) designs and walnut plantations, while root-pruning increased economic returns by extending the period of profitable crop production within the rotation. All agroforestry systems were modelled as having higher returns than monocropping systems (Benjamin *et al.*, 2000).

The effect of grants on profitability and feasibility of agroforestry systems in Europe was explored in a bio-economic model 'FarmSAFE' developed by Graves *et al.* (2007). While silvoarable systems were often the most profitable system (compared to arable and forestry systems) at landscape test sites in France, Spain and the Netherlands under a 'no grants' scenario, a pre-2005 grant regime based on direct area payments, and a post-2005 grant regime associated with the single farm payment scheme changed the profitability of silvoarable systems compared to arable and forestry systems, with some agroforestry systems becoming less profitable (Graves *et al.*, 2007). For example, in the Netherlands, losing arable land for slurry manure application made changing land use to agroforestry uncompetitive.

Palma *et al.* (2007a) used multi-criteria decision analyses to integrate quantitative environmental and economic outputs of agroforestry and allow comparison with conventional agriculture in three European countries. The profitability of the systems varied from country to country depending either on policy or biophysical conditions. In France, analysis indicated that with equal weighting between environmental and economic performance, silvoarable agroforestry was preferable to conventional arable farming, while in Spain and the Netherlands, the overall performance of agroforestry systems depended on the proportion of the farm planted, tree density and land quality used (Palma *et al.*, 2007a).

Recently, there has been considerable interest in placing a monetary value on the delivery of ecosystem services such as soil protection and carbon sequestration. Alavalapati *et al.* (2004) used a 'willingness to pay' approach to identify the economic consequences of internalising non-market goods and services from agroforestry to the benefit of landowners. They found that by including payments for environmental services delivered by agroforestry, the profitability of these systems would increase, relative to conventional agricultural systems.

Based on information on biophysical changes caused by shelterbelts, Kulshreshtha and Kort (2009) estimated the value of external environmental benefits provided by shelterbelt systems in the Canadian prairie provinces as over CDN$140 million. Carbon sequestration accounted for the majority of this (CDN$73 million) and reduced soil erosion also provided significant economic benefits (CDN$15 million). Porter *et al.* (2009) calculated the values of market and non-market ecosystem services of a novel combined food and energy agroforestry system in Taastrup, Denmark. Belts of fast-growing trees (hazel, willow and alder) for bioenergy production were planted at right angles to fields of cereal and pasture crops, and the system was managed organically with no inputs of pesticides or inorganic N. Field-based estimates of ecosystem services including pest control, nitrogen regulation, soil formation, food and forage production, biomass production, soil carbon accumulation, hydrological flow into ground water reserves, landscape aesthetics and pollination by wild pollinators produced a total value of US $1074 ha^{-1}, of which 46% is from market ecosystem services (production of food, forage and biomass crops) and the rest from non-market ecosystem services. Porter *et al.* (2009) then extrapolated these values to the European scale and calculated that the value of non-market ecosystem services from this novel system exceeds current European farm subsidy payments.

There has been considerable interest in the potential of an agroforestry approach to conserve and sequester C while maintaining land for food production and reducing deforestation and degradation of remaining natural forests. The 1997 Kyoto Protocol calls on participating countries to reduce the rising levels of CO_2 and other greenhouse gases by decreasing fossil fuel emissions and accumulating C in soils and vegetation of terrestrial ecosystems. It provides a mechanism by which countries that emit carbon in excess of agreed limits can purchase carbon credits from countries that manage carbon sinks. Leading the way with establishing tradable securities of carbon sinks to off-set emissions, Costa Rica invested $14 million in 1997 for the Payment for Environmental Services (PES), with 80% of funding coming from a tax on fossil fuels and 20% from international sales of carbon permits from public protected areas. This scheme led to the reforestation of 6,500 ha, the sustainable management of 10,000 ha of public natural forests and the preservation of 79,000 ha of private natural forests (Montagnini and Nair, 2004). In 2003, the scheme was expanded to include agroforestry systems, and the Costa Rican government budgeted $400,000 for the integration of agroforestry management into carbon trading schemes with payments depending on the number of trees present on the farm (Oelbermann *et al.*, 2004). Introducing carbon payments to landowners and managers of agroforestry systems in temperate regions opens the way to obtaining additional income from these systems and may increase the attractiveness of establishing an agroforestry system, as well as adding value to established systems such as riparian buffers, shelterbelts, and silvopastoral and silvoarable systems.

4.6.5 Factors affecting the profitability of agroecological approaches

Overall the relative profitability of agroecological farming systems needs to be assessed at the level of the whole farm, rather than the individual gross margins, and should ideally be considered in the long-term, due to the system changes involved, rather than based on the results for one year only. Table 4-10 presents a summary of factors that are likely to impact on farm-level profitability compared with intensive conventional systems.

Table 4-10: Factors affecting the profitability of agroecological compared with intensive conventional systems

Output parameter	***Integrated***	***Organic***	***Agroforestry***	***Key factors***
Output level	0	--	+	Similar yields of main crops in integrated, reduced in organic, multiple outputs in agro-forestry
Value of enterprise mix	0	-	+/-	Increased diversity, need for fertility building on organic farms, availability of markets for specific components
Variable costs	-	--	-	Reduced use of inputs such as fertilisers, pesticides
Fixed costs	-	0/+	+/-	Reduced tillage, higher labour and machinery costs per unit product in organic case, management of tree component
Infrastructure and investments needs	0	+/-	+	Organic may need specialist equipment but also less storage due to lower yields, agroforestry has high establishment costs (specialist equipment for tree component can be supplied by contractors)
Access to premium price markets	+	++	0/+	Most developed for organic, but not all products, certification charges occur
Eligibility for agri-environment support	+	++	[+]	EU Rural Development framework exists but support depends on availability of specific options in schemes in individual countries
Profitability at the farm level	0	0	[+]	Higher prices compensate reduced output and other costs in organic case; agroforestry depends on components

- = less than conventional, 0 = similar to conventional, + = higher than conventional

Source: Own assessment based on literature presented in this section.

All three systems are likely to have some impact on productivity, because of changes in yields when certain inputs are not used and changes in overall output resulting from change in enterprise mix (see Section 4.2). This is offset by increased output from more diverse systems (e.g. timber, livestock products) in agroforestry, whereas the need for the inclusion of fertility-building crops in organic rotations can be a valuable forage resource for livestock but can have a negative impact on profitability if livestock are not present.

Reduced input use leading to changes in direct variable costs was reported for integrated and organic systems. In more diverse systems, however, this can be offset by additional costs for seeds (green manures) and in organic systems higher prices for specialist inputs are common.

Changes in infrastructure and investments arise particularly if new enterprises are introduced, such as in organic farming and agroforestry. These include the need for new machinery, livestock, buildings, fences, and water supplies, feeding systems, manure handling systems, trees and other facilities. This may have impact on working capital requirement and may have impact on fixed costs, such as labour use (see section 4.2). Impacts on fixed costs are likely to be difficult to assess for the range of different types of systems covered by each of three strategies.

Premium prices were so far mainly available for organic products, but in recent years the markets for products produced with high animal welfare standards and other sustainability credentials have grown, which opens up the potential for attaching price premia to products from integrated farming and agroforestry, for example ‘Woodland eggs’. However, developing premium markets requires considerable efforts combined with consumer willingness to pay for specific attributes. Certification charges (for organic and similar schemes) may also be required to achieve premium prices.

The three agroecological strategies are also likely to generate benefits to the environment that might lead to eligibility for agri-environment scheme payments, but any impact on farm profitability will depend on the availability of suitable schemes and the extent to which costs incurred and income foregone are fully compensated. In some cases, payment for ecosystem services (PES) schemes, e.g. by water companies for systems that improve water quality, may be relevant but there are few examples currently in the UK.

The examples presented here show that there is potential for agroecological approaches to achieve similar or even higher profitability than agriculture in general, depending on changes in output levels, cost savings for certain inputs and differences in market prices/access to premium markets if available. It is likely, however, that there will be considerable variability between different farm enterprises. Where market prices are volatile (whether in commodity or premium markets), this can have significant impacts on farmer perceptions of the viability of agroecological approaches, to the extent that the opportunities for input cost savings and risk mitigation through diversification may be overlooked. Decision-making on the viability of agroecological approaches needs to consider all aspects, and needs to be supported by good quality advice and information.

The policy environment, including direct (basic) payments and greening measures, as well as agri-environment, agroforestry and organic farming support options, can make a significant difference to the financial performance of agroecological approaches, or at least to farmer perceptions of the risk involved in their adoption. However, experience with organic farming support measures has shown that adoption will often only occur in contexts where positive market signals and factors stimulating producers to consider conversion (e.g. poor conventional prices or high input costs) operate in tandem. Where there is no stimulus to change, or market signals are weak, policy support alone will not have major impacts, even though profitability levels may be comparable.

4.7 Overview of impacts of key agroecological practices and approaches

The assessment of the different agroecological practices and approaches presented in this Chapter demonstrate that there are differences in performance with respect to each of the objectives, and that there may be both synergies and conflicts between objectives in specific cases. In Table 4-11 we summarise our assessment of the relative contribution of individual practices, as well as of the major approaches (integrated, organic, agroforestry) reviewed. It should be noted that in this table, unlike the earlier ones in this chapter, the scoring represents an assessment of whether the impact is better or worse than conventional intensive systems.

Table 4-11: Contribution of different agroecology practices and approaches to defined sustainable intensification objectives

Practice	***Productivity***	***Non-renewable energy use and GHG emissions***	***Biodiversity and related ecosystem services***	***Soil and water resource protection***	***Profitability***
Fertility-building legume leys	+ (- if not utilised)	+	+ (++ if flowering)	++(if well managed)	-
Organic soil amendments	+	+	++	+	0
Reduced/ zero tillage	+	+	+	+	+
Avoidance of agrochemicals	--	+	++	++	--
Extended crop rotations	+	0/+	+	+	+/-
Polycultures	++	0/+	+	+	+/-
Variety mixtures and populations	+	0/+	+	0	0/-
Field margin and other refugia	+/-	0/+	+/++	0/+	+/-
IPM/biological pest control	+	0/+	+	0	+
Diverse pastures	+	0/+	+	+	0/+
Mixed crops and livestock	+ (if complementary)	0/+	+	+	+/-
Mixed livestock species	+ (if complementary)	0/+	+	0	+/-
Integrated crop/ farm management	0	+	+	+	0/+
Organic farming	--	+ (0 per unit product)	++	++	0 (with premiums)
Agroforestry	+	++	++ (- if bare understorey)	++	+/-

- = worse than conventional, 0 = similar to conventional, + = better than conventional

Source: Own assessment based on literature presented in this section.

Overall, our assessment is that, in general, the potential of agroecological approaches to contribute to sustainable intensification is positive. As discussed above, we recognise that this assessment does not account for sometimes wide performance variations in specific situations. We have also not sought to provide an overall rating combining the different objectives assessed, as the allocation of weightings to individual objectives can vary widely between different stakeholders.

In some cases the impacts could be positive or negative, depending on a) whether the practice, e.g. field margin refugia, enables more cost-savings/yield gain than the land taken out of production, and b) whether the species mixtures used (crops and/or livestock) are complementary and similarly profitable. In some cases, such as the impacts of reduced use of agrochemicals and organic farming on productivity and biodiversity, there is clear evidence of trade-offs that need to be balanced. The resolution of trade-offs is a complex question, which is only starting to be explored in the sustainability literature (e.g. German *et al.*, in review).

Despite the very wide range of studies reviewed in this report, there are still significant methodological challenges to measuring and understanding the relative performance of different practices and approaches. This is the subject of recent and ongoing research (e.g. Defra, 2014a; Vieweger and Doering, 2014) as well as the development of sustainability assessment tools that can be used by farmers and others for monitoring progress and identifying priorities for action in specific situations (e.g. Gerrard *et al.*, 2012; Lampkin *et al.*, 2011; Smith and Little, 2013).

5 CONCLUSIONS AND RECOMMENDATIONS

This report aimed to explore the following key questions:

i. the relationship between the agroecological and the sustainable intensification concepts;
ii. whether agroecological systems and strategies could contribute to sustainable intensification in UK and European contexts;
iii. the extent of any such contribution, i.e. whether agroecological systems can contribute to sustainable intensification as a whole; or whether only some components of agroecological strategies can do so, or whether the two concepts are broadly incompatible;
iv. the extent to which agricultural policy drivers could affect the relationship between sustainable intensification and agroecology;
v. any opportunities and barriers to the wider adoption of whole systems approaches and practices that may form part of agroecological strategies within the UK and Europe

In this Chapter we draw the main conclusions with respect to these questions and make recommendations for future actions.

5.1 The relationship between agroecology and sustainable intensification

In Chapter 2, we explored the definitions of both agroecology and sustainable intensification. In both cases, a wide range of meanings has been attributed to the concepts by different authors and actors.

With respect to sustainable intensification, it has always been clear that the term is broader than simply increasing production, or producing more with less, and that the impact on the environment needs to be considered. However, sources differ on whether this should be limited to no deterioration in the environmental impact, or a positive improvement.

Given the past environmental impacts of agricultural intensification, and the high intensity of much of European agriculture, it can be argued that the enhancement of the environment, and the production of ecosystem services, or non-commodity outputs, should be part of the sustainable intensification process. This could imply, as Buckwell *et al.* (2014) have argued, that the sustainability element should be given a greater emphasis than intensification, but certainly an equal emphasis is appropriate, as BBSRC (2014) has concluded.

The concept of agroecology is used by some in a purely academic sense and by others as a basis for a social/political movement. We have used it here to refer to the use of ecological understanding and principles to support the management and design of agricultural systems. Our definition involves increased consideration of system redesign, rather just input substitution, and is knowledge-based rather than technology-intensive, although technological improvements have not been excluded.

In terms of the relationship between the two concepts, if sustainable intensification includes a strong focus on sustainability and the enhanced provision of environmental goods and ecosystem services, then agroecology has the potential to make a significant contribution to this process. The originator of the sustainable intensification concept (Pretty, 1997) envisaged a strong agroecological underpinning to it. Although this has not been reflected so strongly in subsequent discussions, this report attempts to address this omission. The relevance of agroecology is also reflected in the concepts of eco-functional, or ecological, intensification, which increasingly form part of the sustainable intensification discourse. As the RISE Foundation report (Buckwell *et al.*, 2014) has argued, this also means making knowledge intensification a central focus.

It is questionable whether a sustainable intensification strategy based purely on the more efficient use of technological inputs can deliver the sustainability gains that are needed for the long term, particularly if such inputs are based on the continued use of non-renewable resources (including soil, water and biodiversity where used beyond their regenerative capacity leading to degradation of natural capital). An increased emphasis on agroecological approaches would help address this, including the potential for win-win situations, for example where biodiversity can be enhanced to benefit both wildlife and agricultural production simultaneously.

5.2 The contribution of agroecological systems and strategies

In Chapter 3 we set out to review whether, and how, agroecological practices and systems could contribute to sustainable intensification in UK and European contexts. A wide range of practices, covering the management of soil, plants, animals and people in agriculture, were identified.

The combination of these practices into more or less codified and systematic approaches, including integrated pest and crop management, organic farming, agroforestry and permaculture, were explored. Systematic approaches have potential advantages both in terms of exploiting synergies between components and practices, and the development of markets for products from defined systems. At present, only organic farming has a strongly developed market underpinned by EU-wide regulations. A market focus can present some challenges in fully exploiting the underlying agroecological principles, as the focus of regulations tends to be on the use of auditable inputs, potentially leading to a focus on input substitution rather than on system redesign and environmental outputs.

There are many variants of agroecological systems using terms such as eco-agricultural, ecological farming, low input sustainable farming, regenerative or renewable agriculture. While not as widely adopted as integrated or organic farming, all of these variants share significant common ground. Under the EU Organic Regulations, the terms 'ecological' and 'biological' are defined as synonyms of organic, but this is not always true of the use of these terms in other parts of the world.

While all of the agroecological practices identified can be, and often are, adopted by any producer, including intensive conventional producers, it is both the likelihood of their being adopted (for example, the use of legumes for nitrogen fixation by organic producers) and the systematic framework within which they are used, that makes the difference.

Using specific practices in isolation (such as the avoidance of fertilisers and pesticides) without compensating via other, ideally system redesign-focused, practices can have negative consequences in terms of crop protection and yields. By contrast, combining suitable practices in a systematic way means that the potential for synergistic relationships can be exploited to gain greater benefits. For example, legumes fix nitrogen but they also support pollinators and improve the nutritional and health value of forage crops for livestock.

The adoption of agroecological practices by farmers will also depend on profitability and practicability. Our review illustrates that, while some of the examples involving complex polycultures may be particularly well-suited to labour-intensive, small-scale approaches, others are more applicable to larger-scale systems. For example, the alley systems used in agroforestry ensure that both crop and tree management can be mechanised. Farmers need to assess the costs and benefits of individual practices and systems in the relevant local context before adopting them, but this requires them to have much easier access to the range of agroecological knowledge and information already available.

Sustainable intensification involves more than just technological approaches. New technology undoubtedly has a role to play, and can contribute to improving agroecosystem management, but this needs to be accompanied by the wider adoption of agroecological concepts and knowledge. Such an approach should build on indigenous or tacit farmer knowledge and requires a more participatory approach that involves producers more directly in both agricultural research and innovation. These activities may be less financially rewarding for technology companies and other current market actors and there is a case for joint investment by farmer groups as well as more government intervention so as to increase the supply of public goods.

5.3 Evaluating the extent of the contribution

In Chapter 4, we used available literature and datasets to assess the contribution that different agroecological systems and strategies can make to sustainable intensification. The analysis is complex, partly because of the wide range of relevant outputs and contributory factors, but also because very few studies have made direct comparative evaluations of the type that we have attempted.

In carrying out the assessment, a range of different indicators and metrics used in previous studies were considered. However, there is still a high degree of variability in the kinds of approaches that have been used, with no consensus on a common indicator framework, and some potential key indicators are absent from most studies. For example, Total Factor Productivity (TFP) would have been relevant to this review, but we were unable to locate any studies that had addressed this particular measure on a comparative basis. There is clearly a need to continue with the process of developing sustainability metrics, and in particular to identify those that can build on existing survey datasets as well as the information held on the majority of farms. This will enable both farmers and policy makers to make more effective decisions based on affordable and readily available evidence. The sustainability metrics focus of Defra's Sustainable Intensification Platform (SIP) and the new Agri-tech Strategy's Centre for Informatics and Sustainability Metrics will be relevant in this context.

As this project was a desk study, no primary data collection was undertaken. We therefore relied on a process of expert judgment by the members of the review team. While inevitably subjective to some extent, these judgments were reviewed by a number of external experts and discussed at a seminar in January 2015. Our conclusions take account of the feedback received.

Our evaluation examined five areas of impact: (i) productivity; (ii) energy use and GHG emissions; (iii) biodiversity and related ecosystem services; (iv) soil and water resource use; and (v) profitability.

With respect to **(i) productivity**, we found that there was some potential, as exemplified by integrated crop management, to maintain productivity and increase efficiency/reduce environmental impact through the adoption of agroecological practices such as reduced tillage, rotations/cover crops and the provision of habitats to encourage beneficial insects.

In situations where there was a greater reduction in agrochemical use, and a greater uptake of practices such as reliance on biological nitrogen fixation and soil fertility-building phases in rotations, there was a trade-off involving reduced yields alongside an increased output of environmental goods and associated (non-provisioning) ecosystem services. Despite this, levels of efficiency (in terms of non-renewable resource use and emissions) were not necessarily any worse and often improved.

However, there were also examples of win-win situations, such as within well-designed agroforestry systems and other polycultures, where the total output of commodities and environmental benefits could be increased simultaneously.

In all cases, there was potential for further improvement in productivity through better design and agroecosystem management. As a result, a more relevant question for the debate on sustainable intensification might be 'How can the productivity of each of the different approaches or systems, such as integrated or organic, be enhanced sustainably?' This would contrast with current research approaches focused on the sustainable intensification of conventional production, which may fail to address or exploit the potential benefits arising from adopting a range of different approaches, especially in terms of designing a more resilient food system capable of dealing with a wide range of future potential shocks.

However, we have not been able to specify or quantify the extent of this potential for improvement in a generalisable way given the very wide range of practices and circumstances under which they may be adopted and improved.

With respect to **(ii) energy and GHG emissions, (iii) biodiversity and related ecosystem services (iv) soil and water resource use**, we found that the different agroecological systems reviewed have differing impacts on key environmental and resource use parameters. In general terms, the greater the uptake of agroecological approaches, and in particular the greater the diversity and complexity of agroecosystem components and design, the greater the environmental and resource conservation benefits. However, while there are often financial as well as environmental benefits, there may be trade-offs in terms of labour, management and other financial costs.

With respect to overall **(v) profitability**, the evidence reviewed indicates that integrated crop management systems can maintain comparable or improved profitability compared to intensive conventional systems as a result of the efficiency gains and savings in input costs with similar yields achieved. Agri-environmental support can assist in encouraging specific management practices, in particular the provision of habitats to encourage wildlife while simultaneously generating ecosystem services that support the farming system.

Where yields are substantially reduced, as in organic crop production, then direct support measures and/or specialised, premium markets may be required to maintain comparable levels of profitability, as cost savings will not be sufficient in themselves to compensate for the yield reductions. This is less the case with grass-fed, ruminant livestock production in organic farming, where the output levels and cost savings from reduced fertiliser use are more in balance because the livestock are able to utilise directly the nitrogen-fixing legumes.

The financial returns from agroforestry are more complex to assess, in part because the time frame for income to be generated from the tree component is much longer. This is particularly the case where trees are grown for timber and it may be decades before a return is realised, but even short-term returns from energy, fruit and nut crops can still take several years to be realised. Once established, however, mixed arable and fruit systems for example have the potential to be more profitable as well as more productive than growing such crops separately. The EU Rural Development Regulations provide a framework for supporting the establishment of agroforestry, but this is not yet widely used within the UK.

We have, however, not been able to quantify precisely the specific benefits of individual agroecological approaches in the context of sustainable intensification. It may well be desirable to be able to say 'integrated crop management can achieve an overall improvement in energy use efficiency of wheat production of X% compared with more intensive systems' or 'organic farming can reduce greenhouse gas emissions by Y% per ha against a yield reduction of Z%'. However, the variability within and between the farm types and individual systems may be too great to permit sensible, single-value statements of this type. While further research could attempt to quantify this, it might then imply that both conventional and agroecological systems and practices are static rather than in a state of continuous evolution. It may therefore be better to focus on research designed to improve the delivery of agroecological systems and practices, rather than attempt to precisely quantify the impacts relative to conventional approaches at a particular fixed point in time.

5.4 The role of policy drivers

In the introduction to this report, and at various points throughout, we have identified a series of policy drivers that could potentially affect the further development and the adoption of agroecological approaches in the UK and across the EU.

Within the UK, and in particular England, recent government policy has emphasised the role of research and innovation in driving the uptake of improved technologies and increased production. This is particularly the case with the UK Agri-tech Strategy, but this has been complemented by Defra's Sustainable Intensification Platform (SIP), which emphasises the development of integrated farming systems, interpreted in a broad sense, as well as their market context (see Chapter 1).

At the EU level, a number of relevant policies can be identified, including:

- The Nitrate and Water Framework Directives, which affect nitrate fertiliser and pesticide utilisation in sensitive areas and water catchments;
- The Sustainable Use of Pesticides Directive (see Chapter 1), which has been implemented in the UK via Regulations (2012) and an Action Plan effective from 2014. This requires priority to be given to non-chemical cultural controls and agroecological approaches;
- The latest phase of CAP reform, including the "greening" of direct payments, with its emphasis on crop diversification, protection of permanent grassland and the maintenance of Ecological Focus Areas, together with automatic recognition for organically-certified land;
- Continued provisions under the new round of Rural Development Programmes for agri-environment and climate measures, establishment of agroforestry and organic farming support. However, Defra has decided not to implement the establishment of agroforestry measure, while Wales and Scotland provide limited support;
- Proposals for a new EU Regulation on organic farming with the aim of significantly tightening the rules for organic production, although significant resistance from Member States and others may lead to substantial revisions or withdrawal of the proposals by the new EU Commission;
- The debate on whether or not to permit genetic modification (GM) technologies, if not at EU level then at national levels, which has been under discussion for many years. The use of GMOs is prohibited under EU organic regulations and some Member States plan to use the proposed new national decision making framework to prohibit their use by all producers. Most agroecology advocates are critical of GM technologies, although some argue that there might be opportunities to combine agroecological and GM approaches.
- From a research and innovation perspective, the EU Framework Programme 7 and the new Horizon 2020 programme have contained a number of calls with agroecology, organic farming and conservation agriculture perspectives, as well as an emphasis on multi-actor and participatory involvement of farmers and other businesses;
- The European Innovation Partnership for Agricultural Sustainability (EIP-Agri), supported through the latest round of Rural Development Programmes and likely to be implemented by most UK administrations, provides a mechanism for bringing together operational groups consisting of farmers, advisers, researchers and other businesses in order to develop new research and innovation initiatives. Such a mechanism could be used to support the development of agroecological innovation with producer participation, complementing the Agri-tech Strategy, with its focus more on technology companies and exploitable Intellectual Property.

Since the 1990s, most EU countries, including different parts of the UK at various times, have implemented action plans to support organic farming, but few have focused on agroecology in a more general sense. This is also true of many research programmes, with Wezel *et al.* (2009) commenting on the absence of agroecology research in France and Germany, in contrast with Brazil and the USA. There are some notable exceptions, for example Wageningen University in the Netherlands, (more recently) Coventry University in the UK, as well as some UK research institutes, such as Rothamsted and the John Hutton Institute, which have substantial agroecology groups.

However, the very large German investment in organic farming research through the Bundesprogramm Oekologischer Landbau (BOeLN = Federal Scheme for Ecological (organic) and Sustainable Agriculture) included many agroecological topics. Following an external evaluation of the BOeLN, a decision was taken by the current German government to extend the scope of the programme to other low-input/environmentally friendly production systems, but the resources available have now been significantly reduced. The German support programme also covered a wide range of school and consumer information initiatives as well as producer-focused knowledge transfer programmes.

In 2014, the French government, announced an action plan for agroecology[37] and was a primary sponsor of the FAO conference on Agroecology in Rome in September 2014. Launching the programme in June 2014, the then French Minister of Agriculture, Stéphane Le Foll, said: "*I want our agriculture to go down the road of high performance in terms of both economics and ecology, making the environment a key factor in our competitiveness. This is a dynamic founded on the strength of collective effort and the rich diversity of our regions, on innovation and on the spread of new know-how. We shall make France a leader in agroecology.*"

The French action plan for agroecology *'Producing Differently (Agriculture Produisons Autrement)'* contains 10 key elements:

1. Training current and future farmers, including significant changes to college and university qualifications
2. Promoting a collective dynamic, in particular through the formation of Economic and Environmental Interest Groupings
3. Reducing the use of pesticides, with farmers adopting 'Ecophyto' plans
4. Preferring natural plant protection including biological controls
5. Reducing the use of antibiotics
6. Sustainable development of beekeeping
7. Making good use of livestock effluents
8. Encouraging organic agriculture
9. Choosing and breeding the right seeds
10. Using trees to improve production

The action plan also emphasises the role of farmers in leading the process, including via a network of experimental farms, which have led to a reduction in pesticide use. The French government is keen to drive this process further by means of 'plant protection product saving certificates' which are designed to force distributors to reduce the number of doses used by 20% over a period of five years. This approach, consistent with the issues covered in this report, represents a significant development on the previous French policy of supporting *'Reasoned Agriculture (Agriculture Raisonnée)'* – the French version of integrated crop management.

Key drivers behind the German and French policies are arguably political. The German Bundesprogramm was introduced by the Green Agriculture Minister Renate Kuenast, but has continued to be supported to varying degrees by subsequent coalitions that no longer included the Green Party. In France, the Socialist government has particularly supported the new direction. But they also reflect strong public support for environmental and alternative agricultural policies. These environmental concerns were also reflected in the agenda, not fully realised, of the former EU Agriculture Commissioner Dacian Cioloș.

---

[37] http://agriculture.gouv.fr/IMG/pdf/plaqPA-anglais_vf_cle01abac.pdf see also MAAF (2014a,b)

5.5 Opportunities and barriers to wider adoption

As a result of our analysis, we have identified the following opportunities and barriers to the wider adoption of agroecological approaches:

Conceptualisation of agroecology

There remain a number of potential tensions with respect to the application of agroecology that need to be addressed. During the London project workshop that was organised to discuss the preliminary findings from this report, both contributors and expert reviewers posed a number of questions and challenges. To what extent is agroecology a positivist approach offering a suite of potential solutions as identified in this report, or is it a more normative, social movement advocating a particular direction of travel? Is it possible to operate with global prescriptions or should the emphasis be on local solutions to specific problems? If so how can those solutions be identified? To what extent should the concept be technology-neutral or should it be codified, as in the case of organic farming? Is it always appropriate to mimic nature in finding solutions? Is it realistic to attempt to generate both commodities and non-commodity outputs via a land sharing approach? Is any reduction in output on our most fertile land acceptable? While this report cannot provide definitive answers to any of these questions, there is certainly much potential for debate.

Producer engagement

Experience in other countries indicates that producer engagement is central to successful agroecological strategies. The tradition of engaging producers in the UK is not as strong as it once was, but there are examples such as BASE[38], originally developed in France but now extended to the UK, as well as the Duchy Originals Future Farming Programme (DOFFP) [39] field labs that could be built on.

Knowledge and information

The importance of information in driving enhanced production and environmental outputs was emphasised by Burgess and Morris (2009) as part of the Foresight review. The lack of knowledge about agroecological approaches is a significant issue in the UK. With the possible exception of integrated crop management, agroecology is rarely covered in the agricultural media, and is poorly addressed in most college and university agricultural courses. There is currently no support for training or information on organic agriculture or agroforestry in England, despite the fact that better training and information availability would mean more effective use being made of agri-environmental support for such activities. The Defra-funded organic farming information hub www.ecofarminginfo.org is being extended (with charitable foundation support) to cover the breadth of agroecological initiatives. The French action plan, with its plan to tackle college and university qualifications, rather than just providing occasional courses, demonstrates a willingness to tackle some of the more deep-seated issues.

---

[38] BASE (Biodiversity, Agriculture, Soil and Environment) is the leading network for conservation agriculture in France. Its goal is to facilitate the exchange of information between farmers and agronomists. It is self-funded and independent, run by a board of farmer members. Starting from Brittany, it now has 900 members spread all over France. Originally 'no-till' oriented, it now also covers organic, agroforestry, grazing and BASE-UK (http://base-uk.co.uk/). BASE offers field days, conferences and training to provide practical answers to its members. Topics covered include cover-crop mixes, companion cropping, strip-till etc..

[39] http://www.soilassociation.org/innovativefarming/duchyfuturefarmingprogramme/abouttheprogramme

Research and innovation

There is a need to encourage agroecological research and innovation, in a participatory framework, recognising that a focus on knowledge rather than technology-based research outputs can lead to problems regarding who might fund such work. The BASE and DOFFP models provide examples of agroecologically-orientated, producer-led activities, but there may be a need to reflect on how applied agroecological research can be supported in future, bearing in mind that it supports the provision of significant public goods.

While there is agroecological research taking place at some universities and research institutes in the UK, it is very dispersed and often not well connected to practical on-farm operations. There is a need for greater co-ordination and integration of the publicly-funded research going on in the UK so that more value is derived from it by farming businesses. Current UK research funding mechanisms are not well suited to deliver this. The Agri-Tech Strategy mentions the importance of inclusivity of approaches, but the funding mechanisms emphasise product rather than knowledge-related outcomes and the public-commercial partnering usually (although not exclusively) emphasises proprietary intellectual property (IP). There is a need for these funding mechanisms to be reformed, recognising that the economic and environmental value of knowledge-based, agroecological innovations in agriculture are not captured simply by new saleable products or services. The European Innovation Partnership model within the new EU Rural Development Regulation, providing for operational groups linking farmers, advisers, researchers and other businesses, could provide an opportunity to support this process provided that sufficient resources are designated.

Financial security

For farmers to adopt agroecological systems and practices requires them to have confidence in the financial viability of such approaches. Currently, apart from organic farming, there is very limited information on the financial performance of many agroecological approaches, including agroforestry. There is a clear need for improved financial performance data. In addition, there may be a need for 'risk sharing' by policy or market actors as discussed below.

Market opportunities

Private sector solutions can be developed to support agroecological initiatives, including premium markets as in the case of organic farming, or closer producer-consumer linkages as in consumer-supported agriculture (CSAs). However, there is a danger with market-driven approaches that market priorities then dominate the agroecological principles, leading to reductions in the benefits obtained. It is also questionable whether the full costs of enabling producers to deliver more environmental benefits should be borne by the consumers of their products only, or whether society at large should contribute to supporting what are societal benefits. Inevitably there will also be tensions with the demands from many other consumers for cheap food, which has, to an extent, contributed to the rapid intensification of agriculture and its associated environmental impacts.

Agri-environmental support

Most of the management options under such schemes tend to be focused on specific practices rather than systems-level change. The organic farming and agroforestry support options under the EU Rural Development Regulations are examples of support for systems level change. The mid-tier group schemes in the new English Countryside Stewardship scheme (or the Nature Improvement Area model) could be seen as alternative options for achieving system level change on farms. There is also an opportunity to consider payment for ecosystem services type arrangements, for example in relation to agroforestry for carbon sequestration or organic farming for the protection of water quality in catchment areas.

Institutional structures

Many of the institutions serving agriculture in the UK are resistant to changes in their world view, with a continuing strong focus on technology-driven, production increases and a limited interest in agroecological alternatives. Given the potential implications for human diets and health, and the need to reduce consumption of livestock products, some of the changes required to achieve long-term global food security may be challenging. Change can happen in such structures, but it may be necessary to consider how that process can be actively supported.

One size fits all?

Finally, there is a question of whether a single approach to the sustainable intensification of agricultural production should be favoured, or whether there is perhaps a mosaic of options that could be pursued in different areas. Localised solutions of this type have been identified by Buckwell *et al.* (2014), with Huxham *et al.* (2014) suggesting that the mosaic approach might take the form of:

- intensive agriculture, using a mix of agro-ecological practices, agrochemicals and high-tech technologies (with a minimum level of environmental protection);
- ‘wildlife-friendly’ farming (e.g. organic, high nature value (HNV) farming, etc.) as agroecological systems and practices (+ appropriate high technology inputs);
- other land uses/land sparing for nature, etc., albeit possibly with co-benefits including ecosystem services of use value to agriculture.

The mosaic approach relates also to the land-sharing/land-sparing arguments outlined in section 4.4.5, and the agroecological perspective that farming systems work better where they can generate ecosystem services that help to support them. This implies that a land-sparing concept where commodities alone are produced on some areas of land is inconsistent with an agroecological approach. Concerns were raised by reviewers of this report that nothing should be done to reduce food output on the best land, but if this cannot be sustained, either due to reliance on non-renewable resources, or due to the long-term degradation of soils and the natural capital base, then a land-sharing approach might represent a better option for the long-term even if there is a production penalty involved.

Agroecological approaches can contribute to supporting intensive production systems on the best land, but this will require creating space for functional biodiversity. At the same time it is clear that some landscapes are not well suited to agricultural production and land-sparing is necessary. For example, most tropical rainforest species can only survive within the forest itself, not in the adjacent agriculturally modified habitats. Non-agricultural habitats can be found across Europe and we are not arguing that these should be converted to agricultural use, although some managed woodlands may be amenable to conversion to appropriate variants of agroforestry. There are, however, many high nature value landscapes that have been created through some, albeit extensive, agricultural or pastoral management. In such situations, agroecological approaches could contribute to further enhancing biodiversity and other landscape values, while at the same time supporting the retention of agricultural use to provide an economic return to the farmer and as means of preserving the high nature value generated by previous agricultural activities.

5.6 Recommendations

On the basis of our analysis in this report, we recommend that:

a) Future work on sustainable intensification should place high priority on the sustainability component of the concept, including eco-functional and knowledge intensification, environmental protection and the delivery of ecosystem services;

b) The potential of agroecological approaches to contribute to sustainable intensification (in the sense described above) should be more widely recognised and developed. Agroecology is not just an option for, but an essential component of, sustainable intensification.

c) Realising the potential of agroecology for sustainable intensification also requires the development of appropriate evaluation metrics, both at farm and regional/landscape level, taking account of systems complexity and of different priorities (e.g. water use) in different areas, to support business and policy decision-making,

d) Policies to mitigate the negative impacts of many agricultural inputs, including fertilisers, pesticides, anti-microbials and anti-helminthics, should emphasise agroecological approaches as well as technological or risk management solutions (as in the EU Sustainable Use of Pesticides Directive and the French Action Plan for Agroecology);

e) Agri-environmental, payments for ecosystem services (PES) and market-based policies (e.g. product certification) should be used to encourage the adoption of agroecological approaches. Such policies should not be restricted to supporting a narrow range of approaches such as intensive conventional or integrated crop management – alternatives such as organic farming, agroforestry, permaculture etc. also merit support and development as part of a diverse, multi-strategy (or mosaic) approach;

f) Improved agroecological information and knowledge exchange systems, building on tacit farmer knowledge and active producer participation, should be developed and promoted – to achieve this there is a need for better integration and co-ordination between individuals and organisations working on the subject and the collaborative development of both on-line resources and traditional extension services;

g) Educational provision at vocational skills, further and higher education levels and more widely should include a stronger focus on agroecological practices and systems. In the short term, this issue can be addressed through the provision of targeted support (using the RDP vocational skills measures), but in the longer term a wide range of educational curricula need to be reviewed and updated;

h) Research and innovation policy should include more focus on the development of agroecological approaches, not just their comparative evaluation. Support policies need to facilitate participatory delivery models and address the challenges involved in securing private sector funding for applied research that generates public knowledge not linked to saleable technologies and intellectual property.

6 ABBREVIATIONS USED

ADAS	Agricultural and environment consultancy
AES	Agri-Environment Schemes
AM	Arbuscular mycorrhizal
BASE	Biodiversity, Agriculture, Soil and Environment
BBSRC	Biotechnology and Biological Sciences Research Council
BIS	Department for Business, Innovation and Skills
BOeLN	Bundesprogramm Ökologischer Landbau und andere Formen nachhaltiger Landwirtschaft (German Federal Scheme for Organic and Sustainable Agriculture)
C	Carbon
CAP	Common Agricultural Policy
CH_4	Methane
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
CSA	Consumer/community-supported agriculture
Defra	Department for Environment, Food and Rural Affairs
DOFFP	Duchy Originals Future Farming Programme
DOK trial	Long-term comparison trial of conventional, integrated, organic and biodynamic systems in Switzerland
ECAF	European Conservation Agricultural Federation
EGTOP	Expert Group for Technical Advice on Organic Production
EIA	Ecologically Intensive Agriculture
EIP-Agri	European Innovation Partnership for Agricultural Sustainability
ESR	Efficiency - Substitution – Redesign framework
EU	European Union
EU SCAR	European Union Standing Committee on Agricultural Research
FADN	Farm Accounting Data Network
FAO	Food and Agriculture Organisation (of the United Nations)
FBS	Farm Business Survey
FWAG	Farming and Wildlife Advisory Group
GHG	Greenhouse Gas
GM	Genetic Modification
GMO	Genetically Modified Organisms
GWCT	Game and Wildlife Conservation Trust
GWP	Global Warming Potential
ha	Hectare
HNV	High Nature Value
IAASTD	The International Assessment of Agricultural Knowledge, Science and Technology for Development
IACPA	Integrated Arable Crop Production Alliance
ICLS	Integrated Crop-Livestock Systems

ICM	Integrated Crop Management
IOBC	International Organisation for Biological and Integrated Control
IFM	Integrated Farm Management
IP	Intellectual Property
IPM	Integrated Pest Management
K	Potassium
LCA	Life Cycle Assessment
LEAF	Linking Environment and Farming
LER	Land Equivalent Ratio
LINSA	Learning and Innovation Networks for Sustainable Agriculture
L(E)ISA	Low (External) Input Sustainable Agriculture
LUPG	Land Use Policy Group
MAFF	Ministry of Agriculture, Fisheries and Food (now part of Defra)
N	Nitrogen
NH_3	Ammonia
N_2O	Nitrous oxide
NAP	National Action Plan
NSO	Net System Output
OECD	Organisation for Economic Co-operation and Development
ORC	Organic Research Centre
P	Phosphorus
PDO	Protected Designation of Origin
PES	Payments for Ecosystem Services
PGPR	Plant Growth-Promoting Rhizobacteria
PGI	Protected Geographical Indication
RDP	Rural Development Programme
RISE	Rural Investment Support for Europe (foundation)
SI	Sustainable Intensification
SIP	Sustainable Intensification Research Platform
SMI	Soil Management Initiative
SNH	Scottish Natural Heritage
SOC	Soil Organic Carbon
SOWAP	Soil and Water Protection Project
SRC	Short Rotation Coppice
TFP	Total Factor Productivity
TSG	Traditional Speciality Guaranteed
UK	United Kingdom
UNCTAD	United Nations Conference on Trade and Development
UNESCO	United Nations Educational, Scientific and Cultural Organization
yr	Year
WRAP	Waste and Resources Action Plan

7 REFERENCES

Adams, W.M., 2012. Feeding the next billion: hunger and conservation. *Oryx* 46, 157-158.

ADAS, 2005. Project report OF0357: Organic egg production - a sustainable method for meeting the organic hen's protein requirements. London: Department for the Environment, Food and Rural Affairs.

Adger, W.N., Brown, K., Shiel, R.S., Whitby, M., 1992. Carbon dynamics of land use in Great Britain. *Journal of Environmental Management* 36, 117-133.

Agegnehu, G., Ghizaw, A., Sinebo, W., 2006. Yield performance and land-use efficiency of barley and faba bean mixed cropping in Ethiopian highlands. *European Journal of Agronomy* 25, 202–207.

Akanvou, R.K., Becker, M., Bastiaans, L., Kropff, M.J., 2007. Morpho-physiological characteristics of cover crops for analysis of upland rice production in relay intercropping systems. *Sciences and Nature* 4, 205-216.

Alavalapati, J.R.R., Shrestha, G.A., Stainback, G.A., Matta, J.R., 2004. Agroforestry development: an environmental economic perspective. *Agroforestry Systems* 61, 299-310.

Albrecht, A., Kandji, S.T., 2003. Carbon sequestration in tropical agroforestry. *Agriculture, Ecosystems and Environment* 99(1-3), 15-27.

Allen, J., Davies, T., McCombe, E. (2007) *Report on carbon emissions related to on-farm milk production.* Penkridge: Kite Consulting.

Allton, K., 2006. *Interactions between soil microbial communities, erodibility and tillage practices.* PhD Thesis, Cranfield: Cranfield University.

Alonso, A.M., Guzman, G.J., 2010. Comparison of the efficiency and use of energy in organic and conventional farming in Spanish agricultural systems. *Journal of Sustainable Agriculture* 34(3), 312-338.

Altieri, M.A., 1995. *Agroecology: The science of sustainable agriculture.* 2nd edition. Boulder: Westview Press.

Altieri, M.A., 2000. Agroecology: Principles and strategies for designing sustainable farming systems. In: Uphoff, N., (ed.) *Agroecological Innovations: Increasing Food Production with Participatory Development.* Routledge, 40-46.

Altieri, M.A., 2009. Agroecology, small farm and food sovereignty. *Monthly Review 61(3),* 102-113.

Altieri, M.A., Nicholls, C.I., 2003. Soil fertility management and insect pests: Harmonizing soil and plant health in agroecosystems. *Soil and Tillage Research* 72(2), 203-211.

Andersen, M.K., Hauggaard-Nielsen, H., Weiner, J., Jensen, E.S., 2007. Competitive dynamics in two- and three-component intercrops. *Journal of Applied Ecology,* 44, 545-551.

Anderson, S.H., Udawatta, R.P., Seobi, T., Garrett, H.E., 2009. Soil water content and infiltration in agroforestry buffer strips. *Agroforestry Systems* 75, 5-16.

Andresen, N., 2000. *The foraging pigs. Resource utilisation, interaction, performance and behaviour of pigs in cropping systems.* Uppsala: Swedish University of Agricultural Sciences.

Angus, J.F., Gardner, P.A., Kirkegaard, J.A., Desmarchelier, J.M., 1994. Biofumigation: Isothiocyanates released from brassica roots inhibit growth of the take-all fungus. *Plant and Soil* 162(1), 107-112.

Anon, 2014. *Square Meal: Why we need a new recipe for the future.* London: Food Research Collaboration.

Arden-Clarke, C., Hodges, R.D., 1988. The environmental effects of conventional and organic/biological farming systems. II. Soil ecology, soil fertility and nutrient cycles. *Biological Agriculture and Horticulture* 5**,** 223-287.

Armengot, L., Berner, A., Blanco-Moreno, J.M., Mäder, P., Sans, F.X., 2015. Long-term feasibility of reduced tillage in organic farming. *Agronomy for Sustainable Development* 35, 339-346.

Aronsson, H., Torstensson, G., Bergstrom, L., 2007. Leaching and crop uptake of N, P and K from organic and conventional cropping systems on a clay soil. *Soil Use and Management* 23(1), 71-81.

Auerswald, K., Keinz, M., Fiener, P., 2003. Soil erosion potential of organic versus conventional farming evaluated by USLE modelling of cropping statistics for agricultural districts in Bavaria. *Soil Use and Management* 19, 305-311.

Austad, I., Hauge, L., 2006. Pollarding in western Norway. *1er Colloque Europeen sur les Trognes, 26-28 October*. Vendome, www.maisonbotanique.com.

Azcón-Aguilar, C., Barea, J.M., 1997. Arbuscular mycorrhizas and biological control of soil-borne plant pathogens – an overview of the mechanisms involved. *Mycorrhiza* 6(6), 457-464.

Badgley, C., Moghtader, J., Quintero, E., Zakem, E., Chappell, M.J., Aviles-Vazquez, K., Samulon, A., Perfecto, I., 2007. Organic agriculture and the global food supply. *Renewable Agriculture and Food Systems* 22, 86-108.

Bailey, M.J., Lilley, A.K., Timms-Wilson, T.M., Spencer-Phillips, P.T.N., 2006. *Microbial Ecology of Aerial Plant Surfaces.* Wallingford: CABI Publishing.

Bailey, A., Basford, W., Penlington, N., Park, J., Keatinge, J., Rehman, T., Tranter, R., Yates, C., 2003. A comparison of energy use in conventional and integrated arable farming systems in the UK. *Agriculture, Ecosystems and Environment* 97(1), 241-253.

Balfour, E.B., 1943. *The Living Soil.* London: Faber and Faber.

Bari, M.A., Schofield, N.J., 1991. Effects of agroforestry-pasture associations on groundwater level and salinity. *Agroforestry Systems* 16, 13-31.

Barr, C., Howard, D., Bunce, B., Gillespie, M., Hallam, C. 1991. Changes in hedgerows in Britain between 1984 and 1990. London: Department of the Environment.

Barrett, R.P., Mebrahtu, T., Hanover, J.W., (eds.) 1990. *Black locust: A multipurpose tree species for temperate climates*. Portland, OR: Timber Press.

Basch, G., Geraghty, J., Stret, B., Sturny, W., 2008. No-tillage in Europe: state of the art: Constraints and perspective. *No-till farming systems.Special Publication* (3), 159-168.

Bassler, A.P., Ciszuk, P., Sjelin, K., 2000. Management of laying hens in mobile houses: A review of experiences. *In*: Hermansen, J.E., Lund, V.E., Thuen, E. (Eds.), *Ecological Animal Husbandry in the Nordic Countries*. Tjele, Denmark: Danish Centre for Organic Farming Research.

Batary, P., Matthiesen, T., Tscharntke, T., 2010. Landscape-moderated importance of hedges in conserving farmland bird diversity of organic vs. conventional croplands and grasslands. *Biological Conservation* 143, 2020-2027.

BBSRC, 2014. *Report of the BBSRC working group on sustainable intensification of agriculture.* Swindon: Biotechnology and Biological Sciences Research Council.

Bealey, W.J., Braban, C.F., Theobald, M.R., Famulari, D., Tang, Y.S., Wheat, A., Grigorova, E., Leeson, S., Twigg, M.M., Dragosits, U., Dore, A.J., Sutton, M.A., Nemitz, E., Loubet, B., Robertson, A., Quinn, A.D., Williams, A., Sandars, D.L., Valatin, G., Perks, M., Watterson, D., 2013. Agroforestry systems for ammonia abatement. Final report, Project AC0201. London: Department for Environment, Food and Rural Affairs.

Begon, M., Harper, J.L., Townsend, C.R., 1996. *Ecology: Individuals, Populations and Communities. 3rd edn.* London: Blackwell Science

Bellon, S., Penvern, S., 2014. *Organic Farming - Prototype for Sustainable Agricultures.* Dordrecht, NL: Springer.

Bellon S, Ollivier G., 2012. L'agroecologie en France: l'institutionnalisation d'utopies. In Goulet F, Magda D, Girard N, Hernandez V. (Eds.) *L'agroecologie en Argentine et en France. Regards croises.* Paris: L'Harmattan, 55-90.

Benavides, R., Douglas, G.B., Osoro, K., 2009. Silvopastoralism in New Zealand: Review of effects of evergreen and deciduous trees on pasture dynamics. *Agroforestry Systems* 76, 327-350.

Bengtsson, J., Ahnstrom, J., Weibull, A.C., 2005. The effects of organic agriculture on biodiversity and abundance: A meta-analysis. *Journal of Applied Ecology* 42, 261-269.

Benjamin, T.J., Hoover, W.L., Seifert, J.R., Gillespie, A.R., 2000. Defining competition vectors in a temperate alley cropping system in the midwestern USA 4. The economic return of ecological knowledge. *Agroforestry Systems* 48, 79-93.

Berendsen, R.L., Pieterse, C.M.J., Bakker, P.A.H.M., 2012. The rhizosphere microbiome and plant health. *Trends in Plant Science* 17(8), 478-486.

Berges, S.A., Moore, L.A.S., Isenhart, T.M., Schultz, R.C., 2010. Bird species diversity in riparian buffers, row crop fields, and grazed pastures within agriculturally dominated watersheds. *Agroforestry Systems* 79(1), 97-110.

Berner, A., Hildermann, I., Fliessbach, A., Pfiffner, L., Niggli, U., Mäder, P., 2008. Crop yield and soil fertility response to reduced tillage under organic management. *Soil and Tillage Research* 101, 89-96.

Bernier-Leduc, M., Vanasse, A., Olivier, A., Bussières, D., Maisonneuve, C., 2009. Avian fauna in windbreaks integrating shrubs that produce non-timber forest products. *Agriculture, Ecosystems and Environment* 131, 16-24.

Berry, P., Stockdale, E., Sylvester-Bradley, R., Philipps, L., Smith, K., Lord, E., Watson, C., Fortune, S., 2003. N, P and K budgets for crop rotations on nine organic farms in the UK. *Soil Use and Management* 19(2), 112-118.

Bertholdsson, N.O., 2004. Variation in allelopathic activity over 100 years of barley selection and breeding. *Weed Research* 44(2), 78-86.

Bertholdsson, N.O., 2005. Early vigour and allelopathy–two useful traits for enhanced barley and wheat competitiveness against weeds. *Weed Research* 45(2), 94-102.

Best, L.B., Whitmore, R.C., Booth, G.M., 1990. Use of cornfields by birds during the breeding season: The importance of edge habitat. *American Midland Naturalist* 123, 84-99.

Bhagwat, S.A., Willis, K.J., Birks, H.J.B., Whittaker, R.J., 2008. Agroforestry: A refuge for tropical biodiversity? *Trends in Ecology and Evolution* 23(5), 261-267.

Bhogal, A., Chambers, B., Whitmore, A., Powlson, D., 2006. *The effects of reduced tillage practices and organic material additions on the carbon content of arable soils.* Summary report for Defra Project SP0561. Notts/Harpenden: Rothamsted Research/ADAS.

Bianchi, F.J.J.A., Boonij, C.J.H., Tscharntke, T., 2006. Sustainable pest regulation in agricultural landscapes: a review on landscape composition, biodiversity and natural pest control. *Proceedings Royal Society B* 273, 1715–1727.

Biesmeijer, J.C., Roberts, S.P.M., Reemer, M., Ohlemüller, R., Edwards, M., Peeters, T., Schaffers, A.P., Potts, S.G., Kleukers, R., Thomas, C.D., Settele, J., Kunin, W.E., 2006. Parallel declines in pollinators and insect-pollinated plants in Britain and the Netherlands. *Science* 313(5785), 351-354.

Biodynamic Association, 2014. *What is biodynamics?* Milwaukee, USA: The Biodynamic Association.

Boardman, J., Poesen, J.E., 2006. *Soil erosion in Europe.* Chichester: Wiley.

Boatman, N.,Willis, K., Garrod, G., Powe, N., 2010. *Benefits of Environmental Stewardship to Agricultural Production.* Report for Natural England. York and Newcastle: The Food and Environment Research Agency (FERA) and Newcastle University.

Boer, I.J.M.d., 2003. Environmental impact assessment of conventional and organic milk production. *Livestock Production Science* 80(1-2), 69-77.

Borin, M., Passoni, M., Thiene, M., Tempesta, T., 2009. Multiple benefits of buffer strips in farming areas. *European Journal of Agronomy*. 32(1)103-111.

Boudreau, 2013. Diseases in intercropping systems. *Annual Review of Phytopathology* 51, 499-519.

Boutin, C., Baril, A., Martin, P.A., 2008. Plant diversity in crop fields and woody hedgerows of organic and conventional farms in contrasing landscapes. *Agriculture, Ecosystems and Environment* 123, 185-193.

Brandle, J.R., Hodges, L., Zhou, X.H., 2004. Windbreaks in North American agricultural systems. *Agroforestry Systems* 61, 65-78.

Breeze, T.D., Bailey, A., Balcombe, K.G. Potts, S.G. 2011. Pollination services in the UK: how important are honeybees? *Agriculture, Ecosystems and Environment* 142, 137-143.

Briggs, S., 2008*. Organic Cereal and Pulse Production: A complete guide*. Marlborough: The Crowood Press Ltd.

Bright, A., Brass, D., Clachan, J., Drake, K.A., Joret, A.D., 2011. Canopy cover is correlated with reduced injurious feather pecking in commercial flocks of free-range laying hens. *Animal Welfare* 20, 329-338

Bright, A., Joret, A.D., 2012. Laying hens go undercover to improve production. *Veterinary Record* 170(9), 228-229.

Brittain, C., Kremen, C. and Klein, A.M. 2013. Biodiversity buffers pollination from changes in environmental conditions. *Global Change Biology* 19, 540-547

Brown, R.J., 2014. *Dual biodiversity benefits from legume-based mixtures*. PhD Thesis School of Agriculture, policy and development. Reading: University of Reading.

Brown, J.K.M., 1995. Pathogens' responses to the management of disease resistance genes. *Advances in Plant Pathology* 11, 75–102.

Brownlow, M.J.C., 1994. The characteristics and viability of land-use systems which integrate pig or poultry production with forestry in the UK. *Department of Agriculture*. Reading: University of Reading.

Brownlow, M.J.C., Dorward, P.T., Carruthers, S.P., 2005. Integrating natural woodland with pig production in the United Kingdom: An investigation of potential performance and interactions. *Agroforestry Systems* 64, 251-263.

Bruggen, A.H.C.v., Semenov, A.M., 2000. In search of biological indicators for soil health and disease suppression. *Applied Soil Ecology* 15, 13–24.

Bruinenberg, M.H., Valk, H., Korevaar, H., Struik, P., 2002. Factors affecting digestibility of temperate forages from seminatural grasslands: A review. *Grass and Forage Science* 57, 292-301.

Brussaard, L. de Ruiter, P.C., Brown. G.G., 2007. Soil biodiversity for agricultural sustainability. *Agriculture, Ecosystems and Environment* 121(3), 233-244.

Buckwell, A., Nordang Uhre, A., Williams, A., Polakova, J., Blum, W.E.H., Schiefer, J., Lair, G., Heissenhuber, A., Schiebl, P., Kramer, C., Haber, W., 2014. *The sustainable intensification of European agriculture.* Brussels: RISE Foundation.

Bullock, D.G., 1992. Crop rotation. *Critical Reviews in Plant Sciences* 11(4), 309-326.

Burgess, P.J., 1999. Effects of agroforestry on farm biodiversity in the UK. *Scottish Forestry* 53(1), 24-27.

Burgess, P.J., Incoll, L.D., Corry, D.T., Beaton, A., Hart, B.J., 2005. Poplar (*populus* spp.) growth and crop yields in a silvoarable experiment at three lowland sites in england. *Agroforestry Systems* 63, 157-169.

Burgess, P.J., Incoll, L.D., Hart, B.J., Beaton, A., Piper, R.W., Seymour, I., Reynolds, F.H., Wright, C., Pilbeam, D.J., Graves, A.R., 2003. The impact of silvoarable agroforestry with poplar on farm profitability and biological diversity: Final report to Defra Project AF0105. Silsoe, Beds; Leeds; Cirencester: Cranfield University; University of Leeds; Royal Agricultural College.

Burgess, P.J., Morris, J., 2009. Agricultural technology and land use futures: the UK case. *Land Use Policy* 26S, S222-S229.

Burner, D.M., Carrier, D.J., Beleskey, D.P., Pote, D.H., Ares, A., Clausen, E.C., 2008. Yield components and nutritive value of *robinia pseudoacacia* and *albizia julibrissin* in Arkansas, USA. *Agroforestry Systems* 72, 51-62.

Burner, D.M., Pote, D.H., Ares, A., 2005. Management effects on biomass and foliar nutritive value of *robinia pseudoacacia* and *gleditsia triacanthos* f. *Inermis* in Arkansas, USA. *Agroforestry Systems* 65, 207-214.

Cabaret J., Chylinski C., Duperray F., Meradi S., Evrard C., Bouilhol M., Berrag B., Sallé G., Nicourt C., 2014. Gastro-intestinal parasites for the farmers: why they do what they do? British Society for Parasitology, 52nd Spring meeting, 6-9 April, Cambridge.

Callaway, M., 1992. A compendium of crop varietal tolerance to weeds. *American Journal of Alternative Agriculture* 7(4), 169–180.

Callicott, J.B., 1999. *Beyond the land ethic: more essays in environmental philosophy.* Albany: Sunny Press.

Campiglia, E., Mancinelli, R., Radicetti, E., Marinari, S., 2011. Legume cover crops and mulches: Effects on nitrate leaching and nitrogen input in a pepper crop (*Capsicum annuum* l.). *Nutrient Cycling in Agroecosystems* 89(3), 399-412.

Cannell, M.G.R., Van Noordwijk, M., Ong, C.K., 1996. The central agroforestry hypothesis: The trees must acquire resources that the crop would not otherwise acquire. *Agroforestry Systems* 34, 27-31.

Caplat, J., 2006. *Mise en place et analyse d'une collecte de données agro-environnementale sur les pratiques de l'agriculture biologique*. Paris: Fédération Nationale d'Agriculture Biologique (FNAB).

Carel, T., 2012. *A comparison of earthworm community structure and soil health indicators between conventional and conservation agricultural practices.* Masters Thesis. Newcastle: Newcastle University.

Carisse, O., Dewdney, M., 2002. A review of non-fungicidal approaches for the control of apple scab. *Phytoprotection* 83, 1-29.

Caron, P., Biénabe, E., Hainzelin, E., 2014. Making transition towards ecological intensification of agriculture a reality: The gaps in and the role of scientific knowledge. *Current Opinion in Environmental Sustainability* 8, 44-52.

Carson, R., 1962. *Silent Spring.* Boston: Houghton Mifflin.

Carter, J., Jones, A., O'Brien, M., Ratner, J., Wuerthner, G., 2014. Holistic Management: misinformation on the science of grazed ecosystems. Review article. *International Journal of Biodiversity* Article ID163431, 10pp.

Carvell, C., Meek, W.R., Pywell, R.F., Goulson, D., Nowakowski, M., 2007. Comparing the efficacy of agri-environment schemes to enhance bumble bee abundance and diversity on arable field margins. *Journal of Applied Ecology* 44(1), 29-40.

Carvell, C., Westrich, P., Meek, W.R., Pywell, R.F., Nowakowski, M., 2006. Assessing the value of annual and perennial forage mixtures for bumblebees by direct observation and pollen analysis. *Apidologie* 37(3), 326.

Cassman, K.G. 2008. Scientific challenges underpinning the food-versus-fuel debate. *NABC Report* 20, 171–178.

Cassman, K.G., Dobermann, A., Walters, D.T., 2002. Agroecosystems, nitrogen-use efficiency, and nitrogen management. *AMBIO: A Journal of the Human Environment* 31(2), 132-140.

Cassidy, E.S., West, P.C., Gerber, J.S., Foley, J.A., 2013. Redefining agricultural yields: from tonnes to people nourished per hectare. *Environmental Research Letters* 8, 034015

Cederberg, C., Mattsson, B., 2000. Life cycle assessment of milk production — a comparison of conventional and organic farming. *Journal of Cleaner Production* 8(1), 49-60.

Chamberlain, D.E., Fuller, R.J., Bunce, R.G.H., Duckworth, J.C., Shrubb, M., 2000. Changes in the abundance of farmland birds in relation to the timing of agricultural intensification in England and Wales. *Journal of Applied Ecology* 37, 771-788.

Chamberlain, D.E., Joys, A.C., Johnson, P.J., Norton, L.R., Feber, R.E., Fuller, R.J., 2010. Does organic farming benefit farmland birds in winter? *Biology Letters* 6, 82-84.

Chaparro, J., Sheflin, A., Manter, D., Vivanco, J., 2012. Manipulating the soil microbiome to increase soil health and plant fertility. *Biology and Fertility of Soils* 48(5), 489-499.

Chatterton, J., Graves, A., Audsley, E., Morris, J., Williams, A., 2015. Using systems-based life cycle assessment to investigate the environmental and economic impacts and benefits of the livestock sector in the UK. *Journal of Cleaner Production* 86, 1-8.

Chifflot, V., Rivest, D., Olivier, A., Cogliastro, A., Khasa, D., 2009. Molecular analysis of arbuscular mycorrhizal community structure and spores distribution in tree-based intercropping and forest systems. *Agriculture, Ecosystems and Environment* 131, 32-39.

Chin, K.M., Wolfe, M.S., 1984. The spread of *Erysiphe graminis* f. sp. *hordei* in mixtures of barley varieties. *Plant Pathology* 33, 89-100.

Chu, B., Goyne, K.W., Anderson, S.H., Lin, C.-H., Udawatta, R.P., 2010. Veterinary antibiotic sorption to agroforestry buffer, grass buffer and cropland soils. *Agroforestry Systems* 79(1), 67-80.

Clements, D.R., Weise, S.F., Brown, R., Stonehouse, D.P., Hume, D.J., Swanton, C.J., 1995. Energy analysis of tillage and herbicide inputs in alternative weed management systems. *Agriculture, Ecosystems and Environment* 52(2–3), 119-128.

Clough, Y., Holzschuh, A., Gabriel, D., Purtauf, T., Kleijn, D., Kruess, A., Steffan-Dewenter, I., Tscharntke, T., 2007. Alpha and beta diversity of arthropods and plants in organically and conventionally managed wheat fields. *Journal of Applied Ecology* 44, 804-812.

Clough, Y., Kruess, A., Tscharntke, T., 2007. Local and landscape factors in differently managed arable fields affect the insect herbivore community of a non-crop plant species. *Journal of Applied Ecology* 44, 22-28.

Condron, L.M., Cameron, K.C., Di, H.J., Clough, T.J., Forbes, E.A., McLaren, R.G., Silva, R.G., 2000. A comparison of soil and environmental quality under organic and conventional farming systems in New Zealand. *New Zealand Journal of Agricultural Research* 43, 443-466.

Cook, J.C., Gallagher, R.S., Kaye, J.P., Lynch, J., Bradley, B., 2010a. Optimizing vetch nitrogen production and corn nitrogen accumulation under no-till management. *Agronomy Journal* 102(5), 1491-1499.

Cook, R.J., Baker, K.F., 1983. *The nature and practice of biological control of plant pathogens.* St Paul MN, USA: American Phytopathology Society.

Cook, R., Sitton, J., Haglund, W., 1987. Influence of soil treatments on growth and yield of wheat and implications for control of pythium root rot. *Phytopathology* 77(8), 1192-1198.

Cook, S.K., Turley, D., Spink, J., Drysdale, A., 2000. *LINK Integrated Farming Systems. Volume II: The economic evaluation of input decisions.* Project Report 173. London: Home Grown Cereals Authority.

Cook, S., Clarke, J., Moss, S., Butler-Ellis, C., Stobart, R., Davies, K., 2010. *Managing weeds in arable rotations - a guide.* Stoneleigh: HGCA/Agriculture and Horticulture Development Board.

Cook, S.M., Kahn, Z.R., Pickett, J.A., 2007. The use of push-pull strategies in integrated pest management. *Annual Review of Entomology* 52, 375–400.

Coquil, X., Beguin, P. and Dedieu, B. 2014. Transition to self-sufficient mixed crop-dairy farming systems. *Renewable Agriculture and Food Systems* 29, 195-205.

Cormack, W., Metcalfe, P., 2000. *Energy use in organic farming systems.* Defra Final Project Report London: Department of Environment, Food and Rural Affairs.

Crawford, M., 2010. *Creating a Forest Garden.* Totnes: Green Books.

Creamer, N.G., Bennett, M.A., Stinner, B.R., Cardina, J., Regnier, E.E., 1996. Mechanisms of weed suppression in cover crop-based production systems. *Horticulture Science* 31(3), 410-413.

Crews, T.E., Peoples, M.B., 2005. Can the synchrony of nitrogen supply and crop demand be improved in legume and fertilizer-based agroecosystems? A review. *Nutrient Cycling in Agroecosystems* 72(2), 101-120.

Crowder, D.W., Northfield, T.D., Strand, M.R., Snyder, W.E., 2010. Organic agriculture promotes evenness and natural pest control. *Nature* 466, 109-111.

Crowe, S.R., McAdam, J., 1993. Factors affecting herbage biomass production in a mature tree silvopastoral system. *Agroforestry Forum* 4(3), 14-18.

Curry, N., Kirwan, J., 2014. The role of tacit knowledge in developing networks for sustainable agriculture. *Sociologia Ruralis* 54(3), 341-361.

Cuthbertson, A., McAdam, J., 1996. The effect of tree density and species in carabid beetles in a range of pasture-tree agroforestry systems on a lowland site. *Agroforestry Forum* 7(3), 17-20.

CWS, 2001. Memorandum submitted by CWS Farms Group (F26): Focus of farming practice, organic farming experiments 1989-1997, a summary of key findings. Agriculture Select Committee Report, *Organic Farming,* Appendix 20. London: Parliament.

Dabney, S., Delgado, J., Reeves, D., 2001. Using winter cover crops to improve soil and water quality. *Communications in Soil Science and Plant Analysis* 32(7-8), 1221-1250.

Dabney, S.M., Delgado, J.A., Meisinger, J.J., Schomberg, H.H., Liebig, M.A., Kaspar, T., Mitchell, J., Reeves, W., 2010. Using cover crops and cropping systems for nitrogen management. *In*: Delgado, J., Follett, R. (Eds.), *Advances in Nitrogen Management for Water Quality* Ankeny, Iowa: Soil and Water Conservation Society, 230-281.

Dalgaard, T., Halberg, N., Porter, J.R., 2001. A model for fossil energy use in Danish agriculture used to compare organic and conventional farming. *Agriculture, Ecosystems and Environment* 87, 51-65.

Dawkins, M.S., Cook, P.A., Whittingham, M.J., Mansell, K.A., Harper, A.E., 2003. What makes free-range broiler chickens range? *In situ* measurement of habitat preferences. *Animal Behaviour* 66, 151-160.

Dawson, C.J., Hilton, J., 2011. Fertiliser availability in a resource-limited world: Production and recycling of nitrogen and phosphorus. *Food Policy* 36, Supplement 1(0), S14-S22.

Dawson, J.C., Huggins, D.R., Jones, S.S., 2008. Characterizing nitrogen use efficiency in natural and agricultural ecosystems to improve the performance of cereal crops in low-input and organic agricultural systems. *Field Crops Research* 107(2), 89-101.

Decaens, T., Jimenez, J.J., Gioia, C., Measey, G.J., Lavelle, P. 2006. The values of soil animals for conservation biology. *European Journal of Soil Biology* 42, S23-S38.

Defra, 2009. *Safeguarding our soils - a strategy for England.* London: Department for Environment, Food and Rural Affairs.

Defra, 2013a. *Farm practices survey – Greenhouse gas mitigation practices.* London: Department for Environment, Food and Rural Affairs.

Defra, 2013b. *UK national action plan for the sustainable use of pesticides (plant protection products).* London: Department for Environment, Food and Rural Affairs.

Defra, 2014a. Sustainable Intensification Research Platform (SIP) Projects: (1) Integrated Farm Management, LM0201; (2) Delivering benefits at the landscape scale, LM0302; (3) Markets, drivers and interactions across the food-chain- scoping study - LM0303. London: Department for Environment, Food and Rural Affairs.

Defra, 2014b. *The National Pollinator Strategy: For bees and other pollinators in England.* Bristol: Department for Environment, Food and Rural Affairs.

Defra, 2015. *Countryside Stewardship Manual.* London: Department for Environment, Food and Rural Affairs.

Degré, A., Debouche, C., Verhé, D., 2007. Conventional versus alternative pig production assessed by multicriteria decision analyses. *Agronomy for Sustainable Development* 27(9), 185-195.

Deike, S., Pallutt, B., Christen, O., 2008. Investigations on the energy efficiency of organic and integrated farming with specific emphasis on pesticide use intensity. *European Journal of Agronomy* 28(3), 461-470.

Dennis, P., Shellard, L.J.F., Agnew, R.D.M., 1996. Shifts in arthropod species assemblages in relation to silvopastoral establishment in upland pastures. *Agroforestry Forum* 7(3), 14-17.

Derpsch, R., 2007. No-tillage and conservation agriculture: A progress report. *No-till farming systems*. Bangkok: WASWC Special Publication 7-39.

Devendra, C., 1992. Nutritional potential of fodder trees and shrubs as protein sources in ruminant nutrition. *In*: Speedy, A., Pugliese, P.L. (Eds.), *Legume trees and other fodder trees as protein sources for livestock*. Rome: FAO.

Dhima, K., Vasilakoglou, I., Gatsis, T., Eleftherohorinos, I., 2010. Competitive interactions of fifty barley cultivars with *avena sterilis* and *asperugo procumbens*. *Field Crops Research* 117(1), 90-100.

Diacono, M., Rubino, P., Montemurro, F., 2013. Precision nitrogen management of wheat. A review. *Agronomy for Sustainable Development* 33(1), 219-241.

Diekötter, T.S., Wamser, S., Wolters, V., Birkhofer, K., 2010. Landscape and management effects on structure and function of soil arthropod communities in winter wheat. *Agriculture, Ecosystems and Environment* 137, 108-112.

Dix, M.E., Johnson, R.J., Harrell, M.O., Case, R.M., Wright, R.J., Hodges, L., Brandle, J.R., Schoeneberger, M.M., Sunderman, N.J., Fitzmaurice, R.L., Young, L.J., Hubbard, K.G., 1995. Influences of trees on abundance of natural enemies of insect pests: A review. *Agroforestry Systems* 29, 303-311.

Dixon, R.K., 1995. Agroforestry systems: Sources or sinks of greenhouse gases? *Agroforestry Systems* 31, 99-116.

Dobermann, A., 2012. Getting back to the field. *Nature* 485, 176-177.

Dobbie, K.E., Bruneau P.M.C., Towers, W. (eds.) 2011.T*he State of Scotland's Soil.* Edinburgh: Natural Scotland, Scottish Government.

Dominati.E., Patterson, M., Mackay, A.D., 2010. A framework for classifying and quantifying the natural capital and ecosystem services of soils. *Ecological Economics* 69, 1858-1868.

Doran, J.W., Zeiss, M.R., 2000. Soil health and sustainability: Managing the biotic component of soil quality. *Applied Soil Ecology* 15(1), 3-11.

Döring, T.F., Baddeley, J., Hatch, D., Marshall, A., Pearce, B., Roderick, S., Stobart, R., Storkey, J., Watson, C., Wolfe, M., 2013. *Using legume-based mixtures to enhance the nitrogen use efficiency and economic viability of cropping systems.* Stoneleigh: HGCA/Agriculture and Horticulture Development Board.

Döring, T.F., Knapp, S., Kovacs, G., Murphy, K., Wolfe, M.S., 2011. Evolutionary plant breeding in cereals—into a new era. *Sustainability* 3(10), 1944-1971.

Dosskey, M.G., 2001. Toward quantifying water pollution abatement in response to installing buffers on crop land. *Environmental Management* 28(5), 577-598.

Dou, Z., Fox, R.H., Toth, J.D., 1994. Tillage effect on seasonal nitrogen availability in corn supplied with legume green manures. *Plant and Soil* 162(2), 203-210.

Dougherty, M.C., Thevathasan, N.V., Gordon, A.M., Lee, H., Kort, J., 2009. Nitrate and *escherichia coli* nar analysis in tile drain effluent from a mixed tree intercrop and monocrop system. *Agriculture, Ecosystems and Environment* 131, 77-84.

Dover, J.W. 1997. Conservation headlands: Effects on butterfly distribution and behaviour. *Agriculture, Ecosystems and Environment*, 63(1), 31-49.

Doyle, C., Thomas, T., 2000. Chapter 10: The social implications of agroforestry. *In*: Hislop, A.M., Claridge, J. (eds.), *Agroforestry in the UK. Bulletin 122*. Edinburgh: Forestry Commission, 99-106.

Drewry, J.J., 2006. Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: A Review. *Agriculture, Ecosystems and Environment,* 114**,** 159-169.

Drinkwater, L., Janke, R., Rossoni-Longnecker, L., 2000. Effects of tillage intensity on nitrogen dynamics and productivity in legume-based grain systems. *Plant and Soil* 227(1-2), 99-113.

Dumont, B., Bernués, A., 2014. Editorial: Agroecology for producing goods and services in sustainable animal farming systems. *Animal* 8 (Special Issue), 1201-1203.

Dupraz, C., Liagre, F., 2008. *Agroforesterie.* France Agricole Editions. Paris: Edit Press CEC.

Dupraz, C., Newman, S.M., 1997. Chapter 6. Temperate agroforestry: The European way. *In*: Gordon, A.M., Newman, S.M. (Eds.), *Temperate agroforestry systems*. Wallingford: CABI Publishing, 181-236.

Easterling, W.E., Hays, C.J., Easterling, M.M., Brandle, J.R., 1997. Modeling the effect of shelterbelts on maize productivity under climate change: An application of the Epic model. *Agriculture, Ecosystems and Environment* 61, 163-176.

Ebelhar, S., Frye, W., Blevins, R., 1984. Nitrogen from legume cover crops for no-tillage corn. *Agronomy Journal* 76(1), 51-55.

EBLEX, 2009. *Testing the Water: The English Beef and Sheep Production Environmental Roadmap - Phase 2.* Stoneleigh: EBLEX/Agriculture and Horticulture Development Board.

EC, 1991. Council regulation (EEC No 2092/91) of June 1991 on organic production of agricultrual products and indications referring thereto on agricultural products and foodsuffs. *Official Jounal of the European Communities* L198 (22.7.91),1-15.

EC, 2007. Council Regulation (EC No 834/2007) of June 28 2007 on organic production and labelling of organic products and repealling regulation (EEC) no 2092/91. *Official Journal of the European Communities* L189 (20.7.2007), 1-23.

EC, 2008. Commission Regulation (EC) No 889/2008 laying down detailed rules for the implementation of Council Regulation (EC) No 834/2007 on organic production and labeling of organic products with regard to organic production, labelling and control. *Official Journal of the European Communities*, L 250 51 (18.09.2008), 1-84.

EC, 2009. EU Directive 2009/128/EC of the European Parliament and of the Council of 21 October 2009: Establishing a framework for community action to achieve the sustainable use of pesticides. *Official Jounal of the European Communities* L 309, (24.11.2009), 71–86.

EC, 2012. The implementation of the soil thematic strategy and ongoing activities. Brussels: European Commission (accessed online 17.6.2014).

ECPA, n.d. *Integrated crop management: A training resource.* Brussels: European Crop Protection Association.

Edwards, C.A., Lal, R., Madden, P., Miller, R.H., House, G., 1990. *Research on integrated arable farming and organic mixed farming in the Netherlands. Sustainable Agricultural Systems*. Ankeny, Iowa: Soil and Water Conservation Society. 287-296.

Edwards, S.A., 2003. Intake of nutrients from pasture by pigs. *Proceedings of Nutrition Society* 62, 257-265.

Eekeren, N.v., Bokhorst, J., Brussard, L., 2010. Roots and earthworms under grass, clover and a grass-clover mixture. *Soil Solutions for a Changing World*. World Congress of Soil Science, Brisbane, Australia.

Eggenschwiler, L., Richner, N., Schaffner, D., Jacot, K. 2007. Endangered arable flora: how to conserve and promote it? *Agrarforschung*, 14(5), 206-211.

EGTOP, 2012. *Report on Poultry. EGTOP/4/2012*. Brussels: Commission of the European Communities.

Eichhorn, M.P., Paris, P., Herzog, F., Incoll, L.D., Liagre, F., Mantzanas, K., Mayus, M., Moreno, G., Papanastasis, V.P., Pilbeam, D.J., Pisanelli, A., Dupraz, C., 2006. Silvoarable systems in Europe - past, present and future prospects. *Agroforestry Systems 67*, 29-50.

El-Hage Scialabba, N., Müller-Lindenlauf, M., 2010. Organic agriculture and climate change. *Renewable Agriculture and Food Systems* 25(2), 11.

Elliott, J., Firbank, L.G., Drake, B., Cao, Y., Gooday, R., 2013. *Exploring the concept of sustainable intensification.* UK Nature Conservation Agencies Land Use Policy Group.

Eltun, R., 1995. Comparisons of nitrogen leaching in ecological and conventional cropping systems. *Biological Agriculture and Horticulture* 11, 103-114.

Emden, H.V., Peakall, D., 1996. *Beyond Silent Spring: Integrated pest management and chemical safety.* London: Chapman and Hall Ltd.

Erb, K.-H., Haberl, H., Krausmann, F., Lauk, C., Plutzar, C., Steinberger, J.K., Müller, C., Bondeau, A., Waha, K., Pollack, G., 2009. *Eating the planet. Feeding and fuelling the world sustainably, fairly and humanely – a scoping study.* Social ecology working paper, 116. Vienna: Klagenfurt University.

Erenstein, O., 2003. Smallholder conservation farming in the tropics and sub-tropics: A guide to the development and dissemination of mulching with crop residues and cover crops. *Agriculture, Ecosystems and Environment* 100, 17–37.

Eriksen, J., Petersen, S.O., Sommer, S.G., 2002. The fate of nitrogen in outdoor pig production. *Agronomy* 22, 863-867.

Eriksen, J., Hermansen, J.E., Strudsholm, K., Kristensen, K., 2006. Potential loss of nutrients from different rearing strategies for fattening pigs on pasture. *Soil Use and Management* 22, 256-266.

Ertl, P., Knaus, W., Steinwidder, A., 2013. Biologische Milchviehhaltung ohne Konzentratfuttereinsatz: Auswirkungen auf Tiergesundheit, Leistung und Wirtschaftlichkeit. *In*: Neuhoff, D. *et al.* (eds.), *Ideal und Wirklichkeit - Perspektiven ökologischer Landbewirtschaftung. Beiträge zur 12. Wissenschaftstagung ökologischer Landbau*. Bonn, 524-527.

Esperschütz, J., Gattinger, A., Mäder, P., Schloter, M., Fließbach, A., 2007. Response of soil microbial biomass and community structures to conventional and organic farming systems under identical crop rotations. *FEMS Microbiology Ecology* 61(1), 26-37.

Feber, R.E., Johnson, P.J., Firbank, L.G., Hopkins, A., Macdonald, D.W., 2007. A comparison of butterfly populations on organically and conventionally managed farmland. *Journal of Zoology* 273, 30-39.

Fernández, X., Rodríguez, D., Amoedo, L., García, L., 2012. Co-producing cultural coherency: Impact and potentials of civic food networks in Spain. *10 European IFSA Congress Producing and Reproducing Farming Systems. New modes of organisation for sustainable food systems of tomorrow*. Aarhus.

Finckh, M.R., Mundt, C.C., 1992. Stripe rust, yield and plant competition in wheat cultivar mixtures. *Phytopathology* 82, 905-913.

Finckh, M.R., Wolfe, M.S., 2006. Diversification strategies. *In*: Cooke, B.M., Jones, D.G., Kaye B (eds.) *The epidemiology of plant diseases*. Berlin: Springer.

Firbank, L., Carter, N., Darbyshire, J., Potts, G., (eds.) 1992. *The ecology of temperate cereal fields.* Chichester: Wiley.

Fischer, J., Abson, D., Butsic, V., Chappell, M.J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H.G., Wehrden, H.V., 2013. Land sparing versus land sharing: Moving forward. *Aspects of Applied Biology* 121, 105-107.

Fleischer, K., Streitberger, M., Fartmann, T., 2013. The importance of disturbance for the conservation of a low-competitive herb in mesotrophic grasslands. *Biologia* 68(3), 398-403.

Flysjö, A., Cederberg, C., Henriksson, M., Ledgard, S., 2012. The interaction between milk and beef production and emissions from land use change – critical considerations in life cycle assessment and carbon footprint studies of milk. *Journal of Cleaner Production* 28, 134-142.

Fonts, I., Gea, G., Azuara, M., Ábrego, J., Arauzo, J., 2012. Sewage sludge pyrolysis for liquid production: a review. *Renewable and Sustainable Energy Reviews* 16, 2781-2805.

Foresight, 2011. *The Future of Food and Farming*. London: The Government Office for Science.

Frame, J., 1991. Herbage production and quality of a range of secondary grass species at four rates of fertilizer application. *Grass and Forage Science* 46, 139-151.

Friberg, H., Lagerlof, J., Ramert, B., 2005. Influence of soil fauna on fungal plant pathogens in agricultural and horticultural systems. *Biocontrol Science and Technology* 15, 641-658.

Fuller, R.J., Norton, L.R., Feber, R.E., Johnson, P.J., Chamberlain, D.E., Joys, A.C., Mathews, F., Stuart, R.C., Townsend, M.C., Manley, W.J., Wolfe, M.S., Macdonald, D.W., Firbank, L.G., 2005. Benefits of organic farming to biodiversity vary among taxa. *Biology Letters* 1, 431-434.

Gaba, S.; Bretagnolle, F.; Rigaud, T. and Philippot, L. 2014. Managing biotic interactions for ecological intensification of agroecosystems. *Frontiers in Ecology and Evolution – Agroecology and Land Use Systems* 2, 29.

Gabriel, D., Carver, S.J., Durham, H., Kunin, W.E., Palmer, R.C., Sait, S.M., Stagl, S., Benton, T.G., 2009. The spatial aggregation of organic farming in England and its underlying environmental correlates. *Journal of Applied Ecology* 46, 323-333.

Gabriel, D., Sait, S.M., Hodgson, J.A., Schmutz, U., Kunin, W.E., Benton, T.G., 2010. Scale matters: The impact of organic farming on biodiversity at different spatial scales. *Ecology Letters* 13(7), 858-869.

Gabriel, D., Sait, S.M., Kunin, W.E., Benton, T.G., 2013. Food production vs. biodiversity: Comparing organic and conventional agriculture. *Journal of Applied Ecology* 50, 355-364.

Gadermaier, F., Berner, A., Fliessbach, A., Friedel, J.K., Mäder, P., 2011. Impact of reduced tillage on soil organic carbon and nutrient budgets under organic farming. *Renewable Agriculture and Food Systems* 27(1), 68-80.

Gallai, N., Salles, J., Settele, J., Vaissière, B.E., 2009. Economic valuation of the vulnerability of world agriculture confronted with pollinator decline. *Ecological Economics* 68(3), 810-821.

Gallandt, E.R., Weiner, J., 2007. *Crop–weed competition*. In: *eLS*. Chichester: John Wiley & Sons, Ltd.

Gardi, C., Jeffrey, S. 2009. *Soil biodiversity.* European Commission Joint Research Centre Institute for Environment and Sustainability Report. Luxembourg: Office for Official Publications of the European Communities.

Garforth, C., 2010. Education, training and extension for food producers. *R 16B, Foresight Global Food and Farming Project, London.* Reading: Reading University

Garnett, T., Appleby, M.C., Balmford, A., Bateman, I.J., Benton, T.G., Bloomer, P., Burlingame, B., Dawkins, M., Dolan, L., Fraser, D., Herrero, M., Hoffmann, I., Smith, P., Thornton, P.K., Toulmin, C., Vermeulen, S.J., Godfray, H.C.J., 2013. Sustainable intensification in agriculture: Premises and policies. *Science* 341(6141), 33-34.

Garnett, T., Godfray, H.C.J., 2012. *Sustainable intensification in agriculture.* Oxford: Food Climate Research Network and the Oxford Martin Programme on the Future of Food, University of Oxford.

Garwes, D., 2009. Grazed livestock are good news for the UK. Practice with Science Extract 2: *Reducing Emissions from Livestock.* Stoneleigh: Royal Agricultural Society of England.

Gattinger, A., Muller, A., Haeni, M., Skinner, C., Fliessbach, A., Buchmann, N., Mäder, P., Stolze, M., Smith, P., Scialabba, N.E.-H., 2012. Enhanced top soil carbon stocks under organic farming. *Proceedings of the National Academy of Sciences* 109(44), 18226-18231.

Ge, T., Nie, S.A., Wu, J., Shen, J., Xiao, H.A., Tong, C., Huang, D., Hong, Y., Iwasaki, K., 2011. Chemical properties, microbial biomass, and activity differ between soils of organic and conventional horticultural systems under greenhouse and open field management: A case study. *Journal of Soils and Sediments* 11(1), 25-36.

Geiger, F., de Snoo, G.R., Berendse, F., Guerrero, I., Morales, M.B., Oñate, J.J., Eggers, S., Part, T., Bommarco, R., Bengtsson, J., Clement, L.W., Weisser, W.W., Olszewski, A., Ceryngier, P., Hawro, V., Inchausti, P., Fischer, C., Flohre, A., Thies, C., Tscharntke, T., 2010. Landscape composition influences farm mangement effects on farmland birds in winter: A pan-european approach. *Agriculture, Ecosystems and Environment* 139, 571–577.

German, R.N., Thompson, C.E., Benton, T.G., in review. Relationships among multiple aspects of agriculture's environmental impact and productivity: a meta-analysis to guide sustainable agriculture. Leeds: University of Leeds.

Gerrard, C.L., Smith, L.G., Pearce, B., Padel, S., Hitchings, R., Measures, M., 2012. Public goods and farming. *Farming for food and water security, 10. Sustainable Agriculture Reviews no. 8380*. Dordrecht: Springer, 1-22.

Ghaley, B.B., Porter, J.R., 2013. Emergy synthesis of a combined food and energy production system compared to a conventional wheat (*Triticum aestivum*) production system. *Ecological Indicators* 24(0), 534-542.

Girling, R.D., Döring, T.F., Cousins, J., Creissen, H., Crowley, O., Fish, L., Fradgley, N., Griffiths, S., Haigh, Z., Howlett, S. A., Jones, H.E., Knapp, S., Pearce, B.D., Pearce, H., Snape, J., Stobart, R., Winkler, L.W., Whitley, A. and Wolfe, M.S. 2014 in press. *Adaptive winter wheat populations: development, genetic characterisation and application.* Final Report to Defra. Newbury: Organic Research Centre.

Glen, D., Milsom, N., Wiltshire, C., 1990. Effect of seed depth on slug damage to winter wheat. *Annals of Applied Biology* 117(3), 693-701.

Gliessman, S.R., 1995. Sustainable agriculture: An agroecological perspective. *Advances in Plant Pathology* 11, 45-57.

Gliessmann, S.R., 1998. *Agroecology: The ecology of sustainable food systems.* 2nd edition. CRC Press.

Global Partnership on Nutrient Management, 2010. *Building the foundations for sustainable nutrient management.* United Nations Environment Programme.

Gomiero, T., Paoletti, M.G., Pimentel, D., 2008. Energy and environmental issues in organic and conventional agriculture. *Critical Reviews in Plant Sciences* 27(4), 239-254.

Gosling, P., Shepherd, M., 2005. Long-term changes in soil fertility in organic arable farming systems in England, with particular reference to phosphorus and potassium. *Agriculture, Ecosystems and the Environment* 105(1), 425-432.

Goulding, K.W.T., 2000. Nitrate leaching from arable and horticultural land. *Soil Use and Management* 16, 145-151.

Goulding, K., Jarvis, S., Whitmore, A., 2008. Optimizing nutrient management for farm systems. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363(1491), 667-680.

Goulson, D., 2003. Conserving wild bees for crop pollination. *Journal of Food, Agriculture and Environment* 1, 142-144.

Graves, A.R., Burgess, P.J., Palma, J.H.N., Herzog, F., Moreno, G., Bertomeu, M., Dupraz, C., Liagre, F., Keesman, K., van der Wert, W., Koeffeman de Nooy, A., van den Briel, J.P., 2007. Development and application of bio-economic modelling to compare silvoarable, arable and forestry systems in three European countries. *Ecological Economics* 29, 434-449.

Greaves, M., Marshall, E. 1987. Field margins: definitions and statistics. *Field margins*, 35, 3-10.

Green, R.E., Cornell, S.J., Scharlemann, J.P.W., Balmford, A., 2005. Farming and the fate of wild nature. *Science* 307, 550-555.

Greig-Smith, P., Frampton, G., Hardy, A., 1992. *Pesticides, cereal farming and the environment: The Boxworth project.* HMSO.

Griffiths, J., Phillips, D.S., Compton, S.G., Wright, C., Incoll, L.D., 1998. Responses of slug numbers and slug damage to crops in a silvoarable agroforestry landscape. *Journal of Applied Ecology* 35, 252-260.

Gruner, L., Cabaret, J., 1985. Current methods for estimating parasite populations: Potential and limits to control gastro-intestinal and pulmonary strongyles of sheep on pasture. *Livestock Production Science* 20, 53-70.

Gunnarsson, S., Lerner, H., Bo, A., Nordgren, A., 2011. Meat production, climate change and ethics. Proceedings of the XVth International Congress of the International Society for Animal Hygiene (vol. 1): *Animal Hygiene and Sustainable Livestock Production – Innovations in Hygiene, Nutrition and Housing for Healthy Food from Healthy Animals.* Brno, Czech Republic: Tribun EU SRO.

Gupta, N., Kukal, S.S., Bawa, S.S., Dhaliwal, G.S., 2009. Soil organic carbon and aggregation under poplar based agroforestry system in relation to tree age and soil type. *Agroforestry Systems* 76, 27-35.

Guthman, J., 2004. *Agrarian Dreams: the Paradox of Organic Farming in California.* Berkeley, CA: University of California Press.

Haas, G., Wetterich, F., Köpke, U., 2001. Comparing intensive, extensified and organic grassland farming in southern Germany by process life cycle assessment. *Agriculture, Ecosystems and Environment* 83(1–2), 43-53.

Habte, M., 2006. The roles of abuscular mycorrihizas in plant and soil health. In: *Biological Approaches to Sustainable Soil Systems.* CRC Press, Taylor and Francis Group, 131-147.

Halberg, N., Verschuur, G., Goodlass, G., 2005. Farm level environmental indicators: are they useful? An overview of green accounting systems for European farms. *Agriculture, Ecosystems and Environment* 105, 195-212.

Halberg, N., Hermansen, J.E., Kristensen, I.S., Eriksen, J., Tvedegaard, N., Petersen, B.M., 2010. Impact of organic pig production systems on CO_2 emission, C sequestration and nitrate pollution. *Agronomy for Sustainable Development* 30, 721-731.

Hamza, M.A., Anderson, W.K., 2005. Soil compaction in cropping systems: a review of the nature, causes and possible solutions. *Soil and Tillage Research* 82**,** 121-145.

Hancock, M.H., Wilson, J.D., 2003. Winter habitat associations of seed-eating passerines on Scottish farmland: Extensive surveys highlighted the importance of weedy fodder brassicas, stubbles and open farmland landscapes to declining birds. *Bird Study* 50(2), 116-130.

Hanley, P., 2014. *Eleven.* Victoria, BC: Friesen Press.

Hansen, E.M., Djurhuus, J., 1996. Nitrate leaching as affected by long-term N fertilization on a coarse sand. *Soil Use and Management* 12(4), 199-204.

Harvey, C.A., Gonzalez-Villalobos, J.A., 2007. Agroforestry systems conserve species-rich but modified assemblages of tropical birds and bats. *Biodiversity and Conservation* 16, 2257-2292.

Hassall, M., Hawthorne, A., Maudsley, M., White, P., Cardwell, C. 1992. Effects of headland management on invertebrate communities in cereal fields. *Agriculture, Ecosystems and Environment*, 40, 155-178.

Hathaway-Jenkins, L.J., Sakrabani, R., Pearce, B., Whitmore, A.P., Godwin, R. J., 2011. A comparison of soil and water properties in organic and conventional farming systems in England. *Soil Use and Management* 27(2) 133–142.

Hathaway-Jenkins, L.J., 2011. *The effect of organic farming on soil physical properties infiltration and workability.* PhD Thesis. School of Applied Science. Cranfield: Cranfield University.

Havlin, J.L., Kissel, D.E., Maddux, L.D., Claassen, M.M., Long, J.H., 1990. Crop rotation and tillage effects on soil organic carbon and nitrogen. *Soil Science Society America Journal* 54(2), 448-452.

Hawes, C., Squire, G.R., Hallett, P.D., Watson, C.A., Young, M., 2010. Arable plant communities as indicators of farming practice. *Agriculture, Ecosystems and Environment* 138, 17–26.

Hawkins, B.A., Cornell, H.V., (eds.) 1999. *Theoretical Approaches to Biological Control.* Cambridge: University Press.

Hay, R.K.M., Russell, G., Edwards, T.W., 2000. *Crop production in the East of Scotland.* Edinburgh: SASA, pp 1-61.

Haygarth, P.M., and Ritz, K., 2009. The future of soils and land use in the UK: soil systems for the provision of land-based ecosystem services. *Land Use Policy* 26S, S187–S197.

Herrmann, F., Wedemeyer, R., Liebig, N., Buck, H., Hommes, M., Saucke, H. 2010. *Entwicklung situationsbezogener Strategien zur Vermeidung von Möhrenfliegenschäden auf Praxisbetrieben.* Final Report [Development of a situational on-farm strategy to prevent carrot fly related damage in organic carrots]. Witzenhausen, Germany: Universität Kassel, Fachgebiet Ökologischer Pflanzenschutz.

Herzog, F., Schüepp, C., 2013. Are land-sparing and land-sharing real alternatives for European agricultural landscapes? Rethinking Agricultural Systems in the UK. *Aspects of Applied Biology* 121, 109-116.

Hess, H., Tiemann, T., Noto, F., Carulla, J., Kreuzer, M., 2006. Strategic use of tannins as means to limit methane emissions from ruminant livestock. *International Congress Series* 1293, 4.

HGCA 2014. *Black-grass: solutions to the problem.* Information Sheet 30. Stoneleigh: HGCA/Agriculture and Horticulture Development Board.

Hiddink, G.A., Bruggen, A.H.C.v., Termorshuizen, A.J., Raaijmakers, J.M., Semenov, A.V., 2005. Effect of organic management of soils on suppressiveness to *Gaeumannomyces graminis* var. *Tritici* and its antagonist, *Pseudomonas fluorescens*. *European Journal of Plant Pathology* 113, 417-435.

Hijri, I., Sykorova, Z., Oehl, F., Ineichen, K., Mader, P., Wiemken, A., Redecker, D., 2006. Communities of arbuscular mycorrhizal fungi in arable soils are not necessarily low in diversity. *Molecular Ecology* 15, 2277-2289.

Hill, S.B., 1985. Redesigning the food system for sustainability. *Alternatives*, 12, 32-36.

Hill, S., 2014. Considerations for enabling the ecological redesign of organic and conventional agriculture: A social ecology and psychosocial perspective. *In*: Bellon, S., Penvern, S. (eds.), *Organic farming, Prototype for Sustainable Agricultures*. Dordrecht: Springer, p. 401.

Hilton, S., Bennett, A.J., Keane, G., Bending, G.D., Chandler, D., Stobart, R., Mills, P., 2013, Impact of shortened crop rotation of oilseed rape on soil and rhizosphere microbial diversity in relation to yield decline. *PLOS one* 8(4), e59859.

HM Government, 2013. *UK Strategy for Agricultural Technologies.* London: UK Government Departments for Business, Innovation and Skills; Environment, Food and Rural Affairs; International Development.

Hobbs, P.R., Sayre, K., Gupta, R., 2008. The role of conservation agriculture in sustainable agriculture. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363(1491), 543-555.

Hodgson, J.A., Kunin, W.E., Thomas, C.D., Benton, T.G., Gabriel, D., 2010. Comparing organic farming and land sparing: Optimizing yield and butterfly populations at a landscape scale. *Ecology Letters* 13(11), 1358-1367.

Hodkinson, D.J., Critchley, C.N.R., Sherwood, A.J. 1997. A botanical survey of conservation headlands in Breckland Environmentally Sensitive Area, UK. *1997 Brighton Crop Protection Conference - Weeds*, Conference Proceedings Vols 1-3, 979-984.

Hoehn, P., Tscharntke, T., Tylianakis, J.M., Steffan-Dewenter, I., 2008. Functional group diversity of bee pollinators increases crop yield. *Proceedings of the Royal Society of London Series B - Biological Sciences* 153, 101-107.

Hoeppner, J.W., Entz, M.H., McConkey, B.G., Zentner, R.P., Nagy, C.N., 2006. Energy use and efficiency in two canadian organic and conventional crop production systems. *Renewable Agriculture and Food Systems* 21(1), 60-67.

Hoitink, H.A.J., Boehm, M.J., 1999. Biocontrol within the context of soil microbial communities: a substrate-dependent phenomenon. *Annual Review of Phytopathology* 37, 427-446.

Hole, D.G., Perkins, A.J., Wilson, J.D., Alexander, I.H., Grice, P.V., Evans, A.D., 2005. Does organic farming benefit biodiversity? *Biological Conservation* 122, 113-130.

Holland, J., 2004. The environmental consequences of adopting conservation tillage in Europe: Reviewing the evidence. *Agriculture, Ecosystems and Environment* 103(1), 1-25.

Holmgren, D. 2011. *Permaculture: Principles and Pathways beyond Sustainability*. East Meon: Permanent Publications.

Holzschuh, A., Steffan-Dewenter, I., Kleijn, D., Tscharntke, T., 2007. Diversity of flower-visiting bees in cereal fields: Effects of farming system, landscape composition and regional context. *Journal of Applied Ecology* 44, 41-49.

Hoogmoed, W.B., Klaij, M.C. (eds.) 1994. *Le travail du sol pour une agriculture durable.* Rome: Food and Agriculture Organisation.

Hoppe, G.M., Sibbald, A.R., McAdam, J., Eason, W.R., Hislop, A.M., Teklehaimanot, Z., 1996. The UK national network silvopastoral experiment - a co-ordinated approach to research. *Fourth Congress of the European Society of Agronomy,* Book of Abstracts.

Horneburg, B. and Becker, H.C. 2008. Does regional organic screening and breeding make sense? Experimental evidence from organic outdoor tomato breeding. In: Neuhoff, D., Halberg, N., Alfoeldi, T. *et al.* (eds.) *Cultivating the future based on science - Volume 1: Organic crop production.* Proceedings of the second scientific conference of the International Society of Organic Agriculture Research (ISOFAR), 18–20 June 2008, Modena, Italy. Bonn: ISOFAR, pp 670–673.

Horsted, K., 2006. *Increased foraging in organic layers.* Department of Agroecology. Foulum: Danish Institute of Agricultural Sciences.

Horsted, K., Hermansen, J.E., 2007. Whole wheat versus mixed layer diet as supplementary feed to layers foraging a sequence of different forage crops. *Animal* 1, 575-585.

Howard, A., 1940. *An Agricultural Testament.* London: Oxford University Press.

Huber, D.M., Watson, R.D., 1974. Nitrogen form and plant disease. *Annual Review of Phytopathology* 12, 139-165.

Hutcheon, A., Iles, D., Kendall, D., 2001. Earthworm populations in conventional and integrated farming systems in the Life project (SW England) in 1990–2000. *Annals of Applied Biology* 139(3), 361-372.

Huxham, M., Hartley, S., Pretty, J., Tett, P., 2014. *No dominion over nature: Why treating ecosystems like machines will lead to boom and bust in food supply.* London: Friends of the Earth.

Huyghe, C., De Vliegher, A., van Gils, B., Peters, A., 2014. *Grasslands and herbivore production in Europe and effects of common policies.* Versailles Cedex, France: Éditions Quae.

IFOAM, 2005. *The principles of organic agriculture.* Bonn: International Federation of Organic Agricultural Movements.

Incoll, L.D., Burgess, P.J., Evans, R.J., Corry, D.T., Beaton, A., 1997. Temperate silvoarable agroforestry with poplar. *Agroforestry Forum* 8(3), 12-15.

Isaev, A.S., Nedorezov, L.V., Khlebopros R.G., 1994. The boomerang effect in models of pest population control. In: Conway, G. (ed.) *Pest and Pathogen Control: Strategic, Tactical and Policy Models.* Chichester: Wiley, 29-39.

Ispikoudis, I., Sioliou, K.M., 2005. Cultural aspects of silvopastoral systems. *In*: Mosquera-Losada, M.R., McAdam, J., Rigueiro-Rodríguez, A. (Eds.), *Silvopastoralism and sustainable land management: Proceedings of an international congress on silvopastoralism and sustainable management held in Lugo, Spain, 2004*. Wallingford: CABI Publishing, 319-323.

Isted, R., 2005. Wood-pasture and parkland: Overlooked jewels of the English countryside. *In*: Mosquera-Losada, M.R., McAdam, J., Rigueiro-Rodríguez, A. (Eds.), *Silvopastoralism and sustainable land management*. Wallingford: CABI Publishing, 400-401.

Iverson, A.L., Marin, L.E., Ennis, K.K., Gonthier, D.J., Connor-Barrie, B.T., Remfert, J.L., Cardinale, B.J. and Perfecto, I. 2014. Do polycultures promote win-wins or trade-offs in agricultural ecosystem services? A meta-analysis. *Journal of Applied Ecology* 51 (6) 1593–1602.

Jackson, A., Lampkin, N.H., various years. Organic Farm Incomes in England and Wales. Annual reports to Defra. Aberystwyth: Aberystwyth University.

Jackson, B.M., Wheater, H.S., Mcintyre, N.R., Chell, J., Francis, O.J., Frogbrook, Z., Marshall, M., Reynolds, B., Solloway, I., 2008. The impact of upland land management on flooding: Insights from a multiscale experimental and modelling programme. *Journal of Flood Risk Management* 1(2), 71-80.

Jakobsen, M., 2014. *Organic growing pigs in pasture systems - effect of feeding strategy and cropping system on foraging activity, nutrient intake from the range area amd pig performance.* Masters Thesis, Institute of Agroecology. Foulum: Aarhus University.

Jansen, K., 2000. Labour, livelihoods, and the quality of life in organic agriculture. *Biological Agriculture and Horticulture* 17(3), 247-278.

Janvier, C., Villeneuve, F., Alabouvette, C., Edel-Hermann, V., Mateille, T., Steinberg, C., 2007. Soil health through soil disease suppression: Which strategy from descriptors to indicators? *Soil Biology and Biochemistry* 39(1), 1-23.

Jeffries, P., Gianinazzi, S., Perotto, S., Turnau, K., Barea, J.-M., 2003. The contribution of arbuscular mycorrhizal fungi in sustainable maintenance of plant health and soil fertility. *Biology and Fertility of Soils* 37(1), 1-16.

Jones, C., Basch, G., Baylis, A., Bazzoni, D., Biggs, J., Bradbury, R., Chaney, K., Deeks, L., Field, R., Gómez, J., Jones, R., Jordan, V., Lane, M., Leake, A., Livermore, M., Owens, P., Ritz, K., Sturny, W., Thomas, F., 2006. *Conservation agriculture in europe: An approach to sustainable crop production by protecting soil and water?* Jeallots Hill, Bracknell: SOWAP.

Jordan, N., 1993. Prospects for weed control through crop interference. *Ecological Applications* 3, 84-91.

Jordan, V.W.L., Hutcheon, J.A., 1994. Economic viability of less-intensive farming systems designed to meet current and future policy requirements: 5-year summary of the LIFE project. *Aspects of Applied Biology* 40, 61-67.

Jordan, V., Hutcheon, J. 1995. Less-intensive farming and the environment: an integrated farming systems approach for UK Arable Crop Production. In: Glen, D.M., Greaves, P., Anderson, H.M., (Eds.) *Ecology and integrated farming systems: proceedings of the 13th Long Ashton International Symposium.* Chichester: Wiley, 307-318.

Jordan, V., Leake, A., 2004. Contributions and interactions of cultivations and rotations to soil quality, protection and profitable production. *Managing soil and roots for profitable production*. Stoneleigh: HGCA conference presentation.

Jordan, V., Leake, A., Ogilvy, S., Cook, S., Cormack, W., Green, M., Holland, J., Welsh, J., 2000. Agronomic and environmental implications of soil management practices in integrated farming systems. *Aspects of Applied Biology* 62, 61-66.

Jordan, V.W.L., Leake, A.R., Ogilvy, S., Higginbotham, S., 2000. The economics of integrated farming systems in the UK. *Aspects of Applied Biology* 62, 239-244.

Jørgensen, U., Dalgaard, T., Kristensen, E.S., 2005. Biomass energy in organic farming—the potential role of short rotation coppice. *Biomass and Bioenergy* 28(2), 237-248.

Jose, S., 2009. Agroforestry for ecosystem services and environmental benefits: An overview. *Agroforestry Systems* 76, 1-10.

Jose, S., Gillespie, A.R., Pallardy, S.G., 2004. Interspecific interactions in temperate agroforestry. *Agroforestry Systems* 61, 237-255.

Kaffka, S., Koepf, H.H., 1989. A case study on the nutrient regime in sustainable farming. *Biological Agriculture and Horticulture* 6(2), 89-106.

Karlen, D., Duffy, M., Colvin, T., 1995. Nutrient, labor, energy and economic evaluations of two farming systems in Iowa. *Journal of Production Agriculture* 8(4), 9.

Karlen, D.L., Cambardella, C.A., Kovar, J.L., Colvin, T.S., 2013. Soil quality response to long-term tillage and crop rotation practices. *Soil and Tillage Research* 133, 54-64.

Kassam, A., Friedrich, T., 2009. Nutrient management in conservation agriculture: A biologically-based approach to sustainable production intensification. *7th Conservation Agriculture Conference*. Dnipropetrovsk, Ukraine, 23-26 July 2009, Rome: Food and Agriculture Organization.

Keatinge, J., *et al.,* 1999. *Assessment of the financial and economic impacts demonstrated by low input, integrated farming system experiments.* Report to MAFF for Project CSA 2935. University of Reading, UK.

Keatinge, R., 1996. *Controlling internal parasites without anthelmintics (a review)*. Project OF0132 report to Defra. Newcastle-upon-Tyne: ADAS Redesdale.

Kendall, D., Chinn, N., Smith, B., Tidboald, C., Winstone, L., Western, N., 1991. Effects of straw disposal and tillage on spread of barley yellow dwarf virus in winter barley. *Annals of Applied Biology* 119(2), 359-364.

Kendall, P.E., Nielsen, P.P., Webster, J.R., Verkerk, G.A., Littlejohn, R.P., Matthews, L.R., 2006. The effects of providing shade to lactating dairy cows in a temperate climate. *Livestock Science* 103, 148-157.

Khan, Z.R., Midega, C.A.O., Pittchar, J., Pickett, J.A., Bruce, T.J.A., 2011. Push-pull technology: a conservation agriculture approach for integrated management of insect pests, weeds and soil health in Africa. *International Journal of Agricultural Sustainability* 1, 162-170.

Khusro, M., Andrew, N.R., Nicholas, A., 2012. Insects as poultry feed: A scoping study for poultry production systems in Australia. *World's Poultry Science Association* 68, 435-446.

Kibblewhite, M., Ritz, K., Swift, M., 2008. Soil health in agricultural systems. *Philosophical Transactions of the Royal Society B: Biological Sciences* 363(1492), 685-701.

Kilbride, A.L., Mason, S.A., Honeyman, P.C., Pritchard, D.G., Hepple, S., Green, L.E., 2012. Associations between membership of farm assurance and organic certification schemes and compliance with animal welfare legislation. *Veterinary Record* 170(6), 152.

King, F.H., 1911. *Farmers of Forty Centuries or Permanent Agriculture in China, Korea and Japan*. Madison: King.

King, J.A., Bradley, R.I., Harrison, R., Carter, A.D., 2004. Carbon sequestration and saving potential associated with changes to the management of agricultural soils in england. *Soil Use and Management* 20, 394-402.

Kirchmann, H., Bergström, L., 2001. Do organic farming practices reduce nitrate leaching? *Communications in Soil Science and Plant Analysis* 32(7), 997-1028.

Kirwan, L., Lüscher, A., Sebastia, M., Finn, J., Collins, R., Porqueddu, C., 2007 Evenness drives consistent diversity effects in intensive grassland systems across 28 European sites. *Journal of Ecology* 95, 530–539.

Klaa, K., Mill, P.J., Incoll, L.D., 2005. Distribution of small mammals in a silvoarable agroforestry in Northern England. *Agroforestry Systems* 63, 101-110.

Klerkx, L., van Mierlo, B., Leeuwis, C., 2012. Evolution of systems approaches to agricultural innovation: concepts, analysis and interventions. In: Darnhofer, I., Gibbon, D., Dedieu, B., (eds) *Farming Systems Research into the 21st Century: The New Dynamic.* Dordrecht: Springer, 457-483.

Kloppenburg J., 1991. Social theory and the de/reconstruction of agricultural science: local knowledge for an alternative agriculture. *Rural Sociology* 56, 519-548.

Knight, S., Kightley, S., Bingham, I., Hoad, S., Lang, B., Philpott, H., Stobart, R., Thomas, J., Barnes, A., Ball, B., 2012. *Desk study to evaluate contributory causes of the current 'yield plateau' in wheat and oilseed rape.* Report No. 502, Stoneleigh: HGCA/Agriculture and Horticulture Development Board.

Knudsen, M., Hermansen, J., Halberg, N., Andreasen, L., Williams, A., 2011. Life cycle asssessment of organic food and farming systems: Methodological challenges related to greenhouse gas emissions and carbon sequestration. In: *Organic Agriculture and Climate Change Mitigation.* Report of the Round Table on Organic Agriculture and Climate Change. Rome: Food and Agriculture Organisation.

Koch, A., McBratney, A., Lal, R., 2012. Global soil week: put soil security on the global agenda. *Nature* 492, 186.

Koch, A., McBratney, A., Adams, M., Field, D., Hill, R., Crawford, J., Minasny, B., Lal, R., Abbott, L., O'Donnell, A., 2013. Soil security: Solving the global soil crisis. *Global Policy* 4(4), 434-441.

Koepf, H., Pettersson, B., Schaumann, W., 1976. *Bio-dynamic agriculture: An introduction.* Spring Valley, NY: The Anthroposophic Press.

Kontturi, M., Laine, A., Niskanen, M., Hurme, T., Hyovela, M., Peltonen-Sainio, P., 2011. Pea-oat intercrops to sustain lodging resistance and yield formation in Northern European conditions. *Acta Agriculturae Scandinavica B-Soil and Plant* 61, 612-621.

Köpke, U., 1995. Nutrient management in organic farming systems: The case of nitrogen. *Biological Agriculture and Horticulture* 11(1-4), 15-29.

Korsaeth, A., 2008. Relations between nitrogen leaching and food productivity in organic and conventional cropping systems in a long-term field study. *Agriculture, Ecosystems and Environment* 127(3), 177-188.

Korsaeth, A., Eltun, R., 2008. Nitrogen mass balances in conventional, integrated and ecological cropping systems and the relationship between balance calculations and nitrogen runoff in an 8-year field experiment in Norway. *Agriculture, Ecosystems and Environment* 79(2-3), 199-214.

Koutsouris, A., 2012. Facilitating agricultural innovation systems: A critical realist approach. *Studies in Agricultural Economics* 114, 64-70.

Kragten, S., de Snoo, G.R., 2008. Field-breeding birds on organic and conventional arable farms in the Netherlands. *Agriculture, Ecosystems and Environment* 126, 270-274.

Krauss, M., Berner, A., Burger, D., Wiemken, A., Niggli, U., Mäder, P., 2010. Reduced tillage in temperate organic farming: Implications for crop management and forage production. *Soil Use and Management* 26, 12-20.

Kruse, M., Strandberg, M., Strandberg, B., 2000. *Ecological effects of allelopathic plants: A review.* Technical Reports 315. Silkeborg, DK: Ministry of the Environment and Energy, National Environmental Research Institute.

Kuemmel, B., Langer, V., Magid, J., De Neergaard, A., Porter, J.R., 1998. Energetic, economic and ecological balances of a combined food and energy system. *Biomass and Bioenergy* 15(4-5), 407-416.

Kulshreshtha, S., Kort, J., 2009. External economic benefits and social goods from prairie shelterbelts. *Agroforestry Systems* 75, 39-47.

Kuntz, M., Berner, A., Gattinger, A., Scholberg, J.M., Mäder, P., Pfiffner, L., 2013. Influence of reduced tillage on earthworm and microbial communities under organic arable farming. *Pedobiologia - International Journal of Soil Biology* 56, 251-260.

Kuo, S., Sainju, U., Jellum, E., 1997. Winter cover crop effects on soil organic carbon and carbohydrate in soil. *Soil Science Society of America Journal* 61(1), 145-152.

Küstermann, B., Kainz, M., Hülsbergen, K.J., 2008. Modeling carbon cycles and estimation of greenhouse gas emissions from organic and conventional farming systems. *Renewable Agriculture and Food Systems* 23(1), 38-52.

Lacombe, S., Bradley, R.L., Hamel, C., Beaulieu, C., 2009. Do tree-based intercropping systems increase the diversity and stability of soil microbial communities? *Agriculture, Ecosystems and Environment* 131, 25-31.

Lal, R., 2004. Soil carbon sequestration impacts on global climate change and food security. *Science* 304, 1623-1627.

Lammerts van Bueren, E.T., Myers J.R. (eds.), 2012. *Organic crop breeding.* Chichester: John Wiley and Sons, Inc.

Lampkin, N.H., 1990. *Organic Farming.* Ipswich: Farming Press.

Lampkin, N.H., 1994. Changes in physical and financial performance during conversion to organic farming: Case studies of two English dairy farms. In: Lampkin, N., Padel, S. (Eds.), *The Economics of Organic Farming: An International Perspective*. Wallingford: CABI Publishing, 223-242.

Lampkin, N.H., 2003. Organic farming. In: Soffe, R.J., (ed.) *Primrose McConnell's Agricultural Notebook.* 20th ed. Oxford: Blackwell Science, 288-303.

Lampkin, N.H., 2007. Organic farming's contribution to climate change and agricultural sustainability. Welsh Producer Conference. Builth Wells. Aberystwyth: Organic Centre Wales.

Lampkin, N.H., 2010. Organic farming myths and reality. *World Agriculture* 1(2), 46-53.

Lampkin, N., Bailey, A., Lang, B., Wilson, P., Williams, A., Sandars, D., Fowler, S., Gerrard, C., Moakes, S., Mortimer, S., Nicholas, P., Padel, S., 2011. *The Potential for Extending Economic Farm-level Benchmarking to Environmental and Other Aspects of Farm Performance.* Final report to Defra, Project DO0103. Newbury: Organic Research Centre.

Lampkin, N.H., Gerrard, C.L., Moakes, S., 2014. *Long term trends in the financial performance of organic farms in England and Wales, 2006/07-2011/12.* Newbury: Organic Research Centre.

Lampkin, N.H., Measures, M., Padel, S., (eds.) 2014. *2014 Organic Farm Management Handbook.* Newbury: Organic Research Centre.

Langdale, G.W., Blevins, R.L., Karlen, D.L., McCool, D.K., Nearing, M.A., Skidmore, E.L., Thomas, A.W., Tyler, D.D., Williams, J.R., 1991. Cover crop effects on soil erosion by wind and water. *In*: Hargrove, W. (ed.), *Cover crops for clean water.* Ankeny Soil and Water Conservation Society, 15-22.

Lange, G., Böhm, H., Berendonk, C., 2011. Methoden zur Verbesserung der Vegetations-zusammensetzung in ökologisch bewirtschaftetem Dauergrünland. *In*: Leithold, G., *et al.*, (eds.), *Es geht ums ganze: Forschen im dialog von Wissenschaft und Praxis.* Beiträge zur 11. Wissenschafts-tagung ökologischer Landbau, 15.-18. März 2011. Gießen, DE: Justus-Liebig-Universität, 111-114.

Larkin, R.P., Griffin, T.S., 2007. Control of soilborne potato diseases using brassica green manures. *Crop Protection* 26(7), 1067-1077.

Larkin, R.P., Honeycutt, C.W., Olanya, O.M., Halloran, J., He, Z., 2012. Impacts of crop rotation and irrigation on soilborne diseases and soil microbial communities. In: He, Z., Larkin, R., Honeycutt, W., (eds.), *Sustainable Potato Production: Global case studies.* Dordrecht, NL: Springer, 23-41.

Lawton, J., (Chair) 2010. *Making Space for Nature: A review of England's Wildlife Sites and Ecological Network.* London: Department for Environment, Food and Rural Affairs.

Le groupe de Travail Desertification, 2013. *Agroécologie, une transition vers des modes de vie et de développement viables.* Paroles d'acteurs.

LEAF, undated. What is LEAF's Integrated Farm Management (IFM)? Stoneleigh: Linking Environment and Farming.

Leake, A.R., 1995. Focus on farming practice: an integrated approach to solving crop protection problems in conventional and organic agriculture. *Monographs, British Crop Protection Council*, 171.

Leake, A.R., 1997. An evaluation and comparison of energy resource usage in organic, integrated and conventional farming systems. In: Isart, J., Llerena, J.J. (eds.) *Resource Use in Organic Farming – Proceedings of the 3rd ENOF Workshop.* Barcelona: LEAAM-Agroecología CID-CSCIC, 295-298.

Leake, A.R., 2003. Integrated pest management for conservation agriculture. In: García-Torres, L., Benites, J., Martínez-Vilela, A., Holgado-Cabrera, A., (eds.) *Conservation Agriculture.* Dordrecht, NL: Springer, 271-279.

Lee, K.H., Isenhart, T.M., Schultz, R.C., 2003. Sediment and nutrient removal in an established multi-species riparian buffer. *Journal of Soil and Water Conservation* 58, 1-8.

Lee, K.H., Jose, S., 2003. Soil respiration and microbial biomass in a pecan-cotton alley cropping system in southern USA. *Agroforestry Systems* 58, 45-54.

Leifeld, J., Angers, D.A., Chenu, C., Fuhrer, J., Kätterer, T., Powlson, D.S., 2013. Organic farming gives no climate change benefit through soil carbon sequestration. *Proceedings of the National Academy of Sciences* 110, E984.

Leifeld, J., Fuhrer, J., 2010. Organic farming and soil carbon sequestration: what do we really know about the benefits? *AMBIO: A Journal of the Human Environment,* 14.

Leifert, C., Cooper, J., Wilcockson, S., Butler, G., 2009. The myths about sustainable high yields in conventional farming systems. Conference presentation. Newcastle: Nafferton Ecological Farming Group.

Leinonen, I., Williams, A.G., Wiseman, J., Guy, J., Kyriazakis, I., 2012. Predicting the environmental impacts of chicken systems in the United Kingdom through a life cycle assessment: Broiler production systems. *Poultry Science* 91(1), 8-25.

Lemerle, D., Smith, A., Verbeek, B., Koetz, E., Lockley, P., Martin, P., 2006. Incremental crop tolerance to weeds: A measure for selecting competitive ability in Australian wheats. *Euphytica* 149, 85-95.

Leopold, A., Schwartz, C.W., Finch, R., 1949. *A Sand County Almanac with Essays on Conservation from Round River.* New York: Random House.

Lidfors, L.M., Moran, D., Jung, J., Jensen, P., Castren, H., 1994. Behaviour at calving and choice of calving place in cattle kept in different environments. *Applied Animal Behaviour Science* 42(1), 11-28.

Liebman, M., Mohler, C.L., Staver, C.P., 2001. *Ecological Management of Agricultural Weeds.* Cambridge: University Press.

Lillywhite, R. and Rahn, C., 2005. *Nitrogen UK Report.* Biffaward Programme on Sustainable Resource Use. Wellesbourne: Warwick HRI.

Lillywhite, R.D., *et al.*, in progress. *Development of methodology for assessing the environmental, economic and social characteristics of farming systems*. Draft Final Report to Defra, Project OF03086. Wellesbourne: Warwick HRI.

Lithourgidis, A., Dordas, C., Damalas, C., Vlachostergios, D., 2011. Annual intercrops: An alternative pathway for sustainable agriculture. *Australian Journal of Crop Science* 5, 396-410.

Lobley, M., Butler, A., Reed, M., 2009. The contribution of organic farming to rural development: An exploration of the socio-economic linkages of organic and non-organic farms in England. *Land Use Policy* 26, 723.

Lobley, M., Reed, M., Butler, A., Courtney, P., Warren, M., 2005. *The impact of organic farming on the rural economy in England* (Final Report to Defra for RE0117). Exeter: University of Exeter, Centre for Rural Research.

Lockeretz, W., 1991. Information requirements of reduced chemical production methods. *American Journal of Alternative Agriculture* 6(2), 97-103.

Lockeretz, W.E., 1977. *Agriculture and Energy.* New York: Elsevier.

Lockeretz, W.E., 2007. *Organic Farming: An International History.* Wallingford: CABI Publishing.

Longhurst, K., 2010. *Investigating the conservation implications of using zero-tillage in agricultural systems in the UK.* Masters Thesis. London: University College.

Lorenz, M., Alkhafadji, L., Stringano, E., Nilsson, S., Mueller-Harvey, I., Udén, P., 2014. Relationship between condensed tannin structures and their ability to precipitate feed proteins in the rumen. *Journal of the Science of Food and Agriculture* 94(5), 963-968.

Lotter, D-W., 2003. Organic agriculture. *Journal of Sustainable Agriculture* 21(4), 59-128.

Lovett, D.K., Shalloo, L., Dillon, P., O'Mara, F.P., 2006. A systems approach to quantify greenhouse gas fluxes from pastoral dairy production as affected by management regime. *Agricultural Systems* 88(2–3), 156-179.

Lovett, D.K., Stack, L.J., Lovell, S., Callan, J., Flynn, B., Hawkins, M., O'Mara, F.P., 2005. Manipulating enteric methane emissions and animal performance of late-lactation dairy cows through concentrate supplementation at pasture. *Journal of Dairy Science* 88(8), 2836-2842.

Lu, Y.C., Watkins, K.B., Teasdale, J.R., Abdul-Baki, A.A., 2000. Cover crops in sustainable food production. *Food Reviews International* 16, 121-157.

Luff, M.L., 1996. Use of carabids as environmental indicators in grasslands and cereals. *Annales Zoologici Fennici* 33, 185-195.

Lüscher, A., Finn, J., Connolly, J., Sebastia, M., Collins, R., Fothergill, M., Porqueddu, C., Brophy, C., Huguenin-Elie, M., Kirwan, L., Nyfeler, D., Helgadottir, A., 2008. Benefits of sward diversity for agricultural grasslands. *Biodiversity* 9, 29-32.

Lynch, D., MacRae, R., Martin, R., 2011. The carbon and global warming potential impacts of organic farming: Does it have a significant role in an energy constrained world? *Sustainability* (3), 322-362.

Lynch, J.P., 2007. Roots of the second green revolution. Turner Review No. 14. *Australian Journal of Botany* 55(5), 493-512.

Lynggaard, K., 2006. *The Common Agricultural Policy and organic farming: an institutional perspective on continuity and change.* Wallingford: CABI Publishing.

Macdonald, A., Poulton, P., Howe, M., Goulding, K., Powlson, D., 2005. The use of cover crops in cereal-based cropping systems to control nitrate leaching in SE England. *Plant and Soil* 273(1-2), 355-373.

MacFadyen, S., Gibson, R., Raso, L., Sint, D., Traugott, M., Memmott, J., 2009a. Parasitoid control of aphids in organic and conventional farming systems. *Agriculture, Ecosystems and Environment* 133, 14-18.

MacFadyen, S., Gibson, R.H., Polaszek, A., Morris, R.J., Craze, P.G., Planque, R., Symondson, W.O.C., Memmot, J., 2009b. Do differences in food web structure between organic and conventional farms affect the ecosystem service of pest control? *Ecology Letters* 12, 229-238.

MacLeod, A., Wratten, S.D., Sotherton, N.W., Thomas, M.B., 2004. Beetle banks as refuges for beneficial arthropods in farmland: long-term changes in predator communities and habitat. *Agricultural and Forest Entomology* 6(2), 147-154.

MacMillan, T., Benton, T.G., 2014. Engage farmers in research. *Nature* 508, 25-27.

MacRae, R.J.; Hill, S.B.; Henning, J., Mehuys, G.R., 1989. Farm-scale agronomic and economic conversion from conventional to sustainable agriculture. *Advances in Agronomy* 43, 155-198.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Jossi, W., Widmer, F., Oberson, A., Frossard, E., Oehl, F., Wiemken, A., Gattinger, A., Niggli, U., 2006. The DOK experiment (Switzerland). In: Raup, J., Pekrun, M., Oltmanns, M., Köpke, U. (eds.). *Long-term field experiments in organic farming.* International Society for Organic Agriculture Research (ISOFAR). Berlin: Verlag Dr. Köster, 17.

Mäder, P., Berner, A., Messmer, M., Fliessbach, A., Krauss, M., Dierauer, H., Clerc, M., Koller, M., Meier, M., Schader, C., 2012. Reduzierte Bodenbearbeitung - deutliche Vorteile für Bodenfruchtbarkeit. *Ökologie und Landbau* 162(2), 25-27.

Mäder, P., Fliessbach, A., Dubois, D., Gunst, L., Fried, P., Niggli, U., 2002. Soil fertility and biodiversity in organic farming. *Science* 296(5573), 1694-1697.

MAAF, 2012. Les chiffres. *Alim' Agri* 26. Paris: Ministere de l'Agriculture de l'Agroalimentaire et de la Foret.

MAAF, 2014a. Le plan d'action global pour l'agro-écologie. Paris: Ministere de l'Agriculture de l'Agroalimentaire et de la Foret. http://agriculture.gouv.fr/plan-action-agroecologie.

MAAF, 2014b. Les 10 clés de l'agro-écologie. Paris: Ministere de l'Agriculture de l'Agroalimentaire et de la Foret. http://agriculture.gouv.fr/Les-10-cles-de-l-agro-ecologie.

MAFF, 1998. *Integrated farming - agricultural research into practice.* A report from the Integrated Arable Crop Production Alliance. London: Ministry of Agriculture, Fisheries and Food.

Magurran, A.E., 2004. *Measuring Biological Diversity*. Malden, MA: Blackwell Publishing.

Mahe, T., Portet, F., 2012. *Les enjeux de la production d'agriculture biologique en France*. Centre d'études et de Prospectives, 50. http://www.agreste.agriculture.gouv.fr/IMG/pdf/analyse501207.pdf

Manning, A.D., Gibbons, P., Lindenmayer, D.B., 2009. Scattered trees: A complementary strategy for facilitating adaptive responses to climate change in modified landscapes? *Journal of Applied Ecology* 46, 915-919.

Marley, C., Cook, R., Barrett, J., Keatinge, R., Lampkin, N., 2006 The effects of birdsfoot trefoil (*Lotus corniculatus*) and chicory (*Cichorium intybus*) when compared with perennial ryegrass (*Lolium perenne*) on ovine gastro-intestinal parasite development, survival and migration. *Veterinary Parasitology* 138(3-4), 280-290.

Marriott, E.E., Wander, M.M., 2006. Total and labile soil organic matter in organic and conventional farming systems. *Soil Science Society America Journal* 70, 950–959.

Marsden, T., Smith, E., 2005. Ecological entrepreneurship: Sustainable development in local communities through quality food production and local branding. *Geoforum* 36(4), 440-451.

McAdam, J., 2000. Environmental impacts. *In*: Hislop, M., Claridge, J., (eds.) *Agroforestry in the UK*. Edinburgh: Forestry Commission.

McAdam, J., Sibbald, A. R., Teklehaimanot, Z., Eason, W.R., 2007. Developing silvopastoral systems and their effects on diversity of fauna. *Agroforestry Systems* 70, 81-89.

McAdam, J., Burgess, P.J., Graves, A.R., Rigueiro-Rodríguez, A., Mosquera-Losada, M.R., 2008. Classifications and functions of agroforestry systems in Europe. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R., Rosa, M., (eds.) *Agroforestry in Europe: Current Status and Future Prospects*. Belfast: Springer, 21-42.

McAdam, J., McEvoy, P.M., 2008. The potential for silvopastoralism to enhance biodiversity on grassland farms in Ireland. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R., Rosa, M., (eds.) *Agroforestry in Europe: Current status and Future Prospects.* Belfast: Springer, 343-358.

McAdam, J., Sibbald, A.R., Teklehaimanot, Z., Eason, W.R., 2007. Developing silvopastoral systems and their effects on diversity of fauna. *Agroforestry Systems* 70, 81-89.

McCarthy, J., McCarthy, B., Horan, B., Pierce, K., Galvin, N., Brennan, A., Delaby, L., 2014. Effect of stocking rate and calving date on dry matter intake, milk production, body weight, and body condition score in spring-calving, grass-fed dairy cows. *Journal of Dairy Science* 97(3), 1693-1706.

McCracken, D.I., Tallowin, J.R., 2004. Swards and structure: the interactions between farming practices and bird food resources in lowland grasslands. *Ibis* 146(S2), 108–114.

McEvoy, P.M., McAdam, J., 2005. Woodland grazing in northern Ireland: Effects on botanical diversity and tree regeneration. In: Mosquera-Losada, M.R., McAdam, J., Rigueiro-Rodríguez, A., (eds.) *Silvopastoralism and Sustainable Land Management*. Wallingford: CABI Publishing.

McIntyre, B.D., Herren, H.R., Wakhungu, J., Watson, R.T., 2009. *Agriculture at a Crossroads.* Synthesis Report. International Assessment of Agricultural Knowledge, Science and Technology for Development (IAASTD). Washington DC: Island Press.

McNeely, J.A., Schroth, G., 2006. Agroforestry and biodiversity conservation - traditional practices, present dynamics, and lessons for the future. *Biodiversity and Conservation* 15, 549-554.

Mead, D.J., Willey, R.W., 1980. The concept of a 'land equivalent ratio' and advantages in yields from intercropping. *Experimental Agriculture* 16, 217-228.

Measures, M., 2014. The myths and truths of mob-stocking. *ORC Bulletin* 116, 6-7.

Melfou, K., Thecharopoulos, A., Papanagiotou, E., 2007. Total factor productivity and sustainable agricultural development. *Economics and Rural Development* 3(1), 32-38.

Milus, E.A., Kristensen, K., Hovmøller, M.S., 2009. Evidence for increased aggressiveness in a recent widespread strain of *Puccinia striiformis f. sp. Tritici* causing stripe rust of wheat. *Phytopathology* 99, 89-94.

Maydell, H.-J.v., 1995. Agroforestry in central, northern and eastern Europe. *Agroforestry Systems* 31, 133-142.

Misselbrook, T.H., Chadwick, D.R., Gilhespy, S.L., Chambers, B.J., Smith, K.A., Williams, J., Dragosites, U., 2010. *Inventory of ammonia emissions from UK agriculture 2009.* Projects AM0127,AC0112. London: Department for Environment, Food and Rural Affairs.

Mitlöhner, F.M., Morrow, J.L., Dailey, J.W., Wilson, S.C., Galyean, M.L., Miller, M.F., McGlone, J.J., 2001. Shade and water misting effects on behaviour, physiology, performance and carcass traits of heat-stressed feedlot cattle. *Journal of Animal Science* 79, 2327-2335.

Moakes, S., Lampkin, N., Gerrard, C.L., 2013. Organic Farm Incomes in England and Wales 2011/12. Aberystwyth and Newbury: Aberystwyth University and Organic Research Centre.

Moakes, S., Lampkin, N., Gerrard, C.L., 2014. Organic Farm Incomes in England and Wales 2012/13. Aberystwyth and Newbury: Aberystwyth University and Organic Research Centre.

Mojtahedi, H., Santo, G.S., Hang, A.N., Wilson, J.H., 1991. Suppression of root-knot nematode populations with selected rapeseed cultivars as green manure. *Journal of Nematology* 23(2), 170-174.

Mollison, W., 1990. *Permaculture: A Practical Guide for a Sustainable Future.* Washington DC: Island Press.

Mollison, W., Slay, R.M., 1994. *Introduction to Permaculture*, 2nd ed. Tylagum, Australia: Tagari Publications.

Mondelaers, K., Aertsens, J., Huylenbroeck, G.V., 2009. A meta-analysis of the differences in environmental impacts between organic and conventional farming. *British Food Journal* 111, 1098-1119.

Montagnini, F., Nair, P.K.R., 2004. Carbon sequestration: An underexploited environmental benefit of agroforestry systems. *Agroforestry Systems* 61, 281-295.

Montgomery, D.R., 2007. Soil erosion and agricultural sustainability. *Proceedings Of The National Academy of Sciences* 104**,** 13268-13272.

Moore, S., 1997. Community supported agriculture success story: Making farm to consumer connections. *Acres USA*, 1-21.

Moraine, M., Duru, M., Nicholas, P., Leterme, P., Therond, O., 2014. Farming system design for innovative crop-livestock integration in Europe. *Animal* 8, 1204-1217.

Morgan, K., Murdoch, J., 2000. Organic vs. conventional agriculture: knowledge, power and innovation in the food chain. *Geoforum* 31(2), 159-173.

Morison, D., Hine, R., Pretty, J.N., 2005. Survey and analysis of labour on organic farms in the UK and Republic of Ireland. *International Journal of Agricultural Sustainability* 3(1), 24-43.

Moschitz, H., Tisenkopfs, T., Brunori, G., Home, R., Kunda, I., Sumane, S., 2014. Final report of the SOLINSA project. Frick, CH: Research Institute of Organic Agriculture (FIBL).

Moss, B., 2008. Water pollution by agriculture. *Philosophical Transaction of the Royal Society B* 363, 659-666.

Mundt, C.C., 2002. Use of multiline cultivars and cultivar mixtures for disease management. *Annual Review of Phytopathology* 40, 40.

Mungai, N.W., Motavalli, P.P., Kremer, R.J., Nelson, K.A., 2005. Spatial variation of soil enzyme activities and microbial functional diversity in temperate alley cropping systems. *Biology and Fertility of Soils* 42, 129-136.

Naeem, M., Compton, S.G., Phillips, D.S., Incoll, L.D., 1994. Factors influencing aphids and their parasitoids in a silvoarable agroforestry system. *Agroforestry Forum* 5(2), 20-23.

Nair, P.K.R., 1991. State-of-the-art of agroforestry systems. *Forest Ecology and Management* 45, 1-4.

NEA, 2011. *The UK National Ecosystems Assessment: Synthesis of the Key Findings.* Cambridge: UNEP-WCMC. http://uknea.unep-wcmc.org/.

Nemecek, T., Huguenin-Elie, O., Dubois, D., Gaillard, G., 2005. *Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau.* FAL Schriftenreihe, 58. Reckenholz, CH: Eidgenössische Forschungsanstalt für Agrarökologie und Landbau (FAL).

Newton, A.C., Hackett, C.A., Swanston, J.S., 2008. Analysing the contribution of component cultivars and cultivar combinations to malting quality, yield and disease in complex mixtures. *Journal of the Science of Food and Agriculture* 88, 2142–2152.

Newton, A.C., Flavell, A.J., George, T.S., Leat, P., Mullholland, B., Ramsay, L., Revoredo-Giha, C., Russell, J., Steffenson, B., Swanston, J.S., Thomas, W.T.B., Waugh, R., White, P.J., Bingham, I.J., 2011. Crops that feed the world 4. Barley: a resilient crop? Strengths and weaknesses in the context of food security. *Food Security* 3, 141-178.

Niggli, U., Fließbach, A., Hepperly, P., Scialabba, N., 2009. *Low greenhouse gas agriculture: Mitigation and adaptation potential of sustainable farming systems.* Rome: Food and Agriculture Organisation.

Norton, L.R., Johnson, P.J., Joys, A.C., Stuart, R.C., Chamberlain, D.E., Feber, R.E., Firbank, L.G., Manley, W.J., Wolfe, M.S., Hart, B., Mathews, F., Macdonald, D.W., Fuller, R.J., 2009. Consequences of organic and non-organic farming practices for field, farm and landscape complexity. *Agriculture, Ecosystems and Environment* 129(1-3), 221-227.

Novak, S.M., Fiorelli, J.L., 2009. Greenhouse gases and ammonia emissions from organic mixed crop-dairy systems: A critical review of mitigation options. *Agronomy for Sustainable Development* (30) 22.

Nowak, B., Nesme, T., David, C., and Pellerin, S., 2013. To what extent does organic farming rely on nutrient inflows from conventional farming? *Environmental Research Letters* 8, 044045.

OECD, 2011. *Challenges for Agricultural Research.* Paris: Organisation for Economic Cooperation and Development.

Oehl, F., Sieverding, E., Mäder, P., Dubois, D., Ineichen, K., Boller, T., Wiemken, A., 2004. Impact of long-term conventional and organic farming on the diversity of arbuscular mycorrhizal fungi. *Oecologia* 138(4), 574-583.

Oelbermann, M., Voroney, R.P., Gordon, A.M., 2004. Carbon sequestration in tropical and temperate agroforestry systems: A review with examples from Costa Rica and Southern Canada. *Agriculture, Ecosystems and Environment* 104, 359-377.

Oelofse, M., Jensen, L., Magid, J., 2013. The implications of phasing out conventional nutrient supply in organic agriculture: Denmark as a case. *Organic Agriculture* 3(1), 41-55.

Oelofse, M., Høgh-Jensen, H., Abreu, L.S., Almeida, G.F., El-Araby, A., Hui, Q.Y., de Neergaard, A., 2010. A comparative study of farm nutrient budgets and nutrient flows of certified organic and non-organic farms in China, Brazil and Egypt. *Nutrient Cycling in Agroecosystems* 87, 455-470.

Oerke, E., Dehne, H., Schönbeck, F., Weber, A., 1994. *Crop Production and Crop Protection: Estimated Losses in Major Food and Cash Crops.* Amsterdam: Elsevier.

Offermann, F., Nieberg, H., 2000. *Economic Performance of Organic Farms in Europe.* Hohenheim: University of Hohenheim.

Ogilvy, S., 2000. *Link Integrated Farming Systems: a field-scale comparison of arable rotations.* Volume I: Experimental Work. London: Home Grown Cereals Authority.

Olesen, J., 2009. Organic farming and the challenges of climate change. *Ecology and Farming,* 44.

Ortas, I., 2012. The effect of mycorrhizal fungal inoculation on plant yield, nutrient uptake and inoculation effectiveness under long-term field conditions. *Field Crops Research* 125, 35-48.

Osterburg, B., Runge, T., (eds.) 2007. *Maßnahmen zur Reduzierung von Stickstoffeinträgen in Gewässer - eine wasserschutzorientierte Landwirtschaft zur Umsetzung der Wasserrahmenrichtlinie.* Braunschweig. DE: Forschungsanstalt für Landwirtschaft.

Østergård, H., Finckh, M.R., Fontaine, L., Goldringer, I., Hoad, S.P., Kristensen, K., Lammerts van Bueren, E., Mascher, F., Munk, L., Wolfe, M.S., 2009. Time for a shift in crop production: Embracing complexity through diversity at all levels. *Journal of the Science of Food and Agriculture,* 89, 1439-1445.

Owen, M.D., Zelaya, I.A., 2005. Herbicide-resistant crops and weed resistance to herbicides. *Pest Management Science* 61(3), 301-311.

Oxley, S. 2007. *Clubroot disease of oilseed rape and other brassica crops.* Technical Note 602. Edinburgh: Scotland's Rural College.

Pacini, C., Wossink, A., Giesen, G., Vazzana, C., Huirne, R., 2003. Evaluation of sustainability of organic, integrated and conventional farming systems: a farm and field-scale analysis. *Agriculture, Ecosystems and Environment* 95(1), 273-288.

Padel, S., Niggli, U., Pearce, B., Schlüter, M., Schmid, O., Cuoco, E., Willer, H., Huber, M., Halberg, N., Micheloni, C., 2010. *Implementation Action Plan for Organic Food and Farming Research.* Brussels: Tecnology Platform Organics, International Federation of Organic Agriculture Movements EU Group.

Palma, J., Graves, A.R., Burgess, P.J., Werf, W.V.D., Herzog, F., 2007a. Integrating environmental and economic performance to assess modern silvoarable agroforestry in Europe. *Ecological Economics* 63, 759-767.

Palma, J.H.N., Graves, A.R., Bunce, R.G.H., Burgess, P.J., de Filippi, R., Keesman, K.J., van Keulen, H., Liagre, F., Mayus, M., Moreno, G., Reisner, Y., Herzog, F., 2007b. Modelling environmental benefits of silvoarable agroforestry in Europe. *Agriculture, Ecosystems and Environment* 119, 320-334.

Papanastasis, V.P., Yiakoulaki, M.D., Decandia, M., Dini-Papanastasi, O., 2008. Integrating woody species into livestock feeding in the Mediterranean areas of Europe. *Animal Feed Science and Technology* 140, 1-17.

Parish, D.M.B., Sotherton, N.W., 2004. Game crops and threatened farmland songbirds in Scotland: a step towards halting population declines? *Bird Study* 51(2), 107-112.

Park, J., Newman, S.M., Cousins, S.H., 1994. The effects of poplar (*P. trichocarpa* x *deltoides*) on soil biological properties in a silvoarable system. *Agroforestry Systems* 25, 111-118.

Peichl, M., Thevathasan, N.V., Gordon, A.M., Huss, J., Abohassan, R.A., 2006. Carbon sequestration potentials in temperate tree-based intercropping systems, Southern Ontario, Canada. *Agroforestry Systems* 66, 243-257.

Peng, R.K., Incoll, L.D., Sutton, S.L., Wright, C., Chadwick, A., 1993. Diversity of airborne arthropods in a silvoarable agroforestry system. *Journal of Applied Ecology* 30, 551-562.

Perfecto, I., Vandermeer, J., Wright, A., 2009. *Nature's Matrix: Linking agriculture, conservation and food sovereignty*. London: Earthscan.

Philipps, L., Stopes, C.,1995. The impact of rotational practice on nitrate leaching losses in organic farming systems in the United Kingdom. *Biological Agriculture and Horticulture* 11, 123-134.

Phillips, D.S., Griffiths, J., Naeem, M., Compton, S.G., Incoll, L.D., 1994. Responses of crop pests and their natural enemies to an agroforestry environment. *Agroforestry Forum* 5(2), 14-20.

Pimbert, M.P., (ed.) 2009. *Towards food sovereignty - reclaiming autonomous food systems.* London: International Institute for Environment and Development.

Pimentel, D., 1991. Diversification of biological control strategies in agriculture. *Crop Protection* 10(4), 243-253.

Pimentel, D., Berardi, G., Fast, S., 1983. Energy efficiency of farming systems: Organic and conventional agriculture. *Agriculture, Ecosystems and Environment* 9(4), 359-372.

Pokarzhevskii, A.D., Zaboyev, D.P., Ganin, G.N., Gordienko, S.A., 1997. Amino acids in earthworms: Are earthworms ecosystemivorous? *Soil Biology and Biochemistry* 29(3/4), 559-567.

Ponder, F., Jones, J.E., Mueller, R., 2005. Using poultry litter in black walnut management. *Journal of Plant Nutrition* 28, 1355-1364.

Ponisio, L.C., M'Gonigle, L.K., Mace, K.C., Palomino, J., de Valpine, P., Kremen, C., 2015. Diversification practices reduce organic to conventional yield gap. *Proceedings of the Royal Society B* 282, 20141396.

Ponti, T., Rijk, B., van Ittersum, M.K., 2012. The crop yield gap between organic and conventional agriculture. *Agricultural Systems* 108, 1-9.

Porter, J., Costanza, R., Sandhu, H., Sigsgaard, L., Wratten, S., 2009. The value of producing food, energy and ecosystem services within an agro-ecosystem. *Ambio* 38(4), 186-193.

Potts, G., 1992. The environmental and ecological importance of cereal fields. In: Firbank, L.G., Carter, N., Darbyshire, J.F., Potts, G.R., (ed.) *The Ecology of Temperate Cereal Fields.* Chichester: Wiley, 3-21.

Potts, S.G., Biesmeijer, J.C., Kremen, C., Neumann, P., Schweiger, O., Kunin, W.E., 2010. Global pollinator declines: trends, impacts and drivers. *Trends in Ecology and Evolution* 25(6), 345-353.

Pretty, J., 1997. The sustainable intensification of agriculture. *Natural Resources Forum* 21, 247-256.

Pretty, J., 1998. *The Living Land: Agriculture, food and community regeneration in rural Europe.* London: Earthscan.

Pretty, J., Toulmin, C., Williams, S., 2011. Sustainable intensification in African agriculture. *International Journal of Agricultural Sustainability* 9(1), 5-24.

Price, G.W., Gordon, A.M., 1999. Spatial and temporal distribution of earthworms in a temperate intercropping system in Southern Ontario, Canada. *Agroforestry Systems* 44, 141-149.

Puckett, H.L., Brandle, J.R., Johnson, R.J., Blankenship, E.E., 2009. Avian foraging patterns in crop field edges adjacent to woody habitat. *Agriculture, Ecosystems and Environment* 131, 9-15.

Pulleman, M., Jongmans, A., Marinissen, J., Bouma, J., 2003. Effects of organic versus conventional arable farming on soil structure and organic matter dynamics in a marine loam in the Netherlands. *Soil Use and Management* 19**,** 157-165.

Pywell, R., Warman, E., Hulmes, L., Hulmes, S., Nuttall, P., Sparks, T., Critchley, C., Sherwood, A., 2006. Effectiveness of new agri-environment schemes in providing foraging resources for bumblebees in intensively farmed landscapes. *Biological Conservation* 129(2), 192-206.

Quested, T., Marsh, E., Stunell, D., Parry, A., 2013. Spaghetti soup: the complex world of food waste behaviours. *Resources, Conservation and Recycling* 79, 43-51.

Rands, M., Sotherton, N., 1986. Pesticide use on cereal crops and changes in the abundance of butterflies on arable farmland in England. *Biological Conservation* 36(1), 71-82.

Ranells, N., Wagger, M., 1997. Winter grass-legume bicultures for efficient nitrogen management in no-till corn. *Agriculture, Ecosystems and Environment* 65, 23-32.

Raper, R.L., 2005. Agricultural traffic impacts on soil. *Journal of Terramechanics* 42**,** 259-280.

Rauch, P., March, S., Brinkmann, J., Spiekers, H., Pries, M., Edmunds, B., Harms, J., 2012. Verbundprojekt Gesundheit und Leistung in der ökologischen Milchviehhaltung - Ansätze in der Fütterung. In: Wiesinger, K., Clais, K., (eds.) *Angewandte Forschung und Beratung für den ökologischen Landbau in Bayern.* Proceedings of Öko-Landbau-Tag 2012, Schriftenreihe der LfL 4/2012. Freising-Weihenstefan: Bayerische Landesanstalt für Landwirtschaft (LfL), 43-49.

Raupp, J., Oltmanns, M., Pekrun, C., Köpke, U., 2006. Soil properties, crop yield and quality with farmyard manure with and without biodynamic preparations and with inorganic fertilizers. In: Raupp, J., Pekrun, C., Oltmanns, M., Köpke, U., (eds.) *Long-term field experiments in organic farming. International Society of Organic Agriculture Research (ISOFAR).* Berlin: Verlag Dr HJ Köster, 135-155.

Ravindren, V., Blair, R., 1993. Feed resources for poultry production in Asia and the Pacific. Iii. Animal protein sources. *World's Poultry Science Association* 49, 219-235.

Reganold, J.P., 2012. The fruits of organic farming. *Nature* 485, 176.

Reganold, J.P., Elliott, L.F., Unger, Y.L., 1987. Long-term effects of organic and conventional farming on soil erosion. *Nature* 330, 370-372.

Reganold, J.P., Palmer, A.S., Lockhart, J.C., Macgregor, A.N., 1993. Soil quality and financial performance of biodynamic and conventional farms in New Zealand. *Science* 260(5106), 344-349.

Reguieg, M.M., Labdi, M., Benbelkacem, A., Hamou, M., Maatougui, M.E.H., Grando, S., Ceccarelli, S., 2013. First experience on participatory barley breeding in Algeria. *Journal of Crop Improvement* 27, 469–486.

Reith, C.C., Guidry, M.J., 2003. Eco-efficiency analysis of an agricultural research complex. *Journal of Environmental Management* 68(3), 219-229.

Rigueiro-Rodríguez, A., Fernández-Núnez, E., Gonzalez-Hernandez, M.P., McAdam, J., Mosquera-Losada, M.R., 2008. Agroforestry systems in Europe: Productive, ecological and social perspectives. In: Rigueiro-Rodríguez, A., McAdam, J., Mosquera-Losada, M.R., Rosa, M., (eds.) *Agroforestry in Europe: Current Status and Future Prospects*. Belfast: Springer, 43-65.

Rillig, M.C., Wright, S.F., Eviner, V.T., 2002. The role of arbuscular mycorrhizal fungi and glomalin in soil aggregation: Comparing effects of five plant species. *Plant and Soil* 238, 325-333.

Robertson, G.P., Paul, E.A., Harwood, R.R., 2000. Greenhouse gases in intensive agriculture: Contributions of individual gases to the radiative forcing of the atmosphere. *Science* 289 (5486), 1922-1925.

Robinson D.A., Hockley, N., Cooper, D.M., Emmett, B.A., Keith, A.M., Lebron, I., Reynolds, B., Tipping, E., Tye, A.M., Watts, C.W., Whalley, W.R., Black, H.I.J., Warren, G.P., 2013. Natural capital and ecosystem services: developing an appropriate soils framework as a basis for valuation. *Soil Biology and Biochemistry* 57, 1023-1033.

Roschewitz, I., Gabriel, D., Tscharntke, T., Thies, C., 2005. The effects of landscape complexity on arable weed species diversity in organic and conventional farming. *Journal of Applied Ecology* 42, 873-882.

Ross, S., Topp, K., Ennos, R., Chagunda, M., 2014. *Breeding, feeding and management to reduce the emissions intensity of dairy production.* Rural Policy Centre Research Briefing 2014/06. Aberdeen: Scotland's Rural College.

Royal Society, 2009. *Reaping the benefits: Science and the sustainable intensification of global agriculture.* London: The Royal Society.

Rundlöf, M., Bengtsson, J., Smith, H.G., 2008. Local and landscape effects of organic farming on butterfly species richness and abundance. *Journal of Applied Ecology* 45, 813-820.

Rundlöf, M., Smith, H.G., 2006. The effect of organic farming on butterfly diversity depends on landscape context. *Journal of Applied Ecology* 43, 1121-1127.

Ryden, J.C., Garwood, E., 1984. Evaluating the nitrogen balance of grassland. In: Hardcastle, J.E.Y., (ed.) *Grassland Research Today*. Swindon: Agricultural and Food Research Council, 10-11.

Ryschawy, J., Choisis, N., Choisis, J.P., Joannon, A., Gibon, A., 2012. Mixed crop-livestock systems: an economic and environmental-friendly way of farming? *Animal* 6, 1722-1730.

Salomon, E., Åkerhielm, H., Lindahl, C., Lindgren, K., 2007. Outdoor pig fattening at two Swedish organic farms – spatial and temporal load of nutrients and potential environmental impact. *Agriculture, Ecosystems and Environment* 121(4), 407-418.

Sans, F.X., Berner, A., Armengot, L., Mäder, P., 2011. Tillage effects on weed communities in an organic winter wheat–sunflower–spelt cropping sequence. *Weed Research* 51(4), 413-421.

Sappok, M., Pellikaan, W., Schenkel, H., Sundrum, A., 2008. *Einsatz von Raufuttermitteln (Silage, Weidelgras, Topinambur und Stoppelrüben) im Vegetationsverlauf in der ganzjährigen Freilandhaltung von Mastschweinen*. Final report BÖL study FKZ 03OE407. Witzenhausen: University of Kassel.

Sarwar, M., Kirkegaard, J.A., Wong, P.T.W., Desmarchelier, J.M., 1998. Biofumigation potential of brassicas. *Plant and Soil* 201(1), 103-112.

Sattler, F., Wistinghausen, E., 1992. *Bio-dynamic Farming Practice.* Stourbridge: Biodynamic Agricultural Association.

Saunders, H., Cook, S., Cormack, W., Green, M., Holland, J., Leake, A., Welsh, J., 2000. Bird species as indicators to assess the impact of integrated crop management on the environment: A comparative study. *Aspects of Applied Biology*(62), 47-53.

Savory, A., 2013. *Response to request for information on the "science" and "methodology" underpinning Holistic Management and holistic planned grazing.* Savory Institute www.savoryinstitute.com.

Savory, A., Butterfield, J., 1999. *Holistic Management: A New Framework for Decision Making.* 2nd Edition. Washington DC: Island Press.

SCAR, 2011. *Sustainable food consumption and production in a resource-constrained world: The 3rd SCAR Foresight Exercise.* Brussels: European Comission, Standing Committee on Agricultural Research.

SCAR, 2012. *Agricultural knowledge and innovation systems in transition - a reflection paper.* Brussels: European Comission, Standing Committee on Agricultural Research.

Schader, C., 2009. *Cost-effectiveness of organic farming for achieving environmental policy targets in Switzerland.* PhD thesis, Aberystwyth University. Frick, Switzerland: Research Institute of Organic Agriculture (FiBL).

Schader, C., Stolze, M., Gattinger, A., 2012. Environmental performance of organic farming. In: Boye, J.I., Arcand, Y. (eds.) *Green Technologies in Food Production and Processing.* New York: Springer, 183-210.

Schädler, M., Brandl, R., Kempel, A., 2010. "Afterlife" effects of mycorrhisation on the decomposition of plant residues. *Soil Biology and Biochemistry* 42, 521-523.

Schmidt, M., Tscharntke, T., 2005. The role of perennial habitats for central European farmland spiders. *Agriculture, Ecosystems and Environment* 105 (1-2), 235-242.

Schmidt, M.H., Roschewitz, I., Thies, C., Tscharntke, T., 2005. Differential effects of landscape and management on diversity and density of ground-dwelling farmland spiders. *Journal of Applied Ecology* 42, 281-287.

Schoeneberger, M.M., 2009. Agroforestry: Working trees for sequestering carbon on agricultural lands. *Agroforestry Systems* 75, 27-37.

Schroeder, P., 1994. Carbon storage benefits of agroforestry systems. *Agroforestry Systems* 27, 89-97.

Schroth, G., Balle, P., Peltier, R., 1995. Alley cropping groundnut with *Gliricidia sepium* in Cote d'Ivoire: Effects on yields, microclimate and crop diseases. *Agroforestry Systems* 29, 147-163.

Schultz, B., Zimmermann, O., Liebig, N., Wedemeyer, R., Leopold, J., Rademacher, J., Katz, P., Rau, F., Saucke, H., 2010. *Anwendung natürlich vorkommender Gegenspieler der Kohlmottenschildlaus (KMSL) in Kohlgemüse im kombinierten Einsatz mit Kulturschutznetzen.* [Application of naturally occurring antagonists of the cabbage white fly (Aleyrodes proletella) in organic crops in combination with netting.] Witzenhausen: University of Kassel.

Schutter, O.d., 2010. *Report Submitted by the Special Rapporteur on the Right to Food.* A/HRC/16/49. United National General Assembly, Human Rights Coucil. http://www2.ohchr.org/english/issues/food/docs/A-HRC-16-49.pdf.

Scialabba, N., Hattam, C. (eds.) 2002. *Organic Agriculture, Environment and Food Security.* Environment and Natural Resources Series, 4. Rome: Food and Agriculture Organisation.

Scialabba, N., Müller-Lindenlauf, M., 2010. Organic agriculture and climate change. *Renewable Agriculture and Food Systems* 25(02), 158-169.

Scialabba, N., Pacini, C., Moller, S., 2014. *Smallholder ecologies*. Rome: Food and Agriculture Organization.

Scott, R., Sullivan, W.C., 2007. A review of suitable companion crops for black walnut. *Agroforestry Systems* 71, 185-193.

Seiter, S., Ingham, E.R., William, R.D., 1999. Dynamics of soil fungal and bacterial biomass in a temperate climate alley cropping system. *Applied Soil Ecology* 12(2), 139-147.

Seufert, V., 2012. *There's nothing black or white about organic agriculture.* Kutztown, PA: Rodale Institute http://rodaleinstitute.org.

Seufert, V., Ramankutty, N., Foley, J.A., 2012. Comparing the yields of organic and conventional agriculture. *Nature* 485(7397), 229-232.

Sexton, P., Plant, A., Johnson, S.B., Jemison, J., 2007. Effect of a mustard green manure on potato yield and disease incidence in a rainfed environment. *Crop Management* 6(1), 63-70.

Sharrow, S.H., Ismail, S., 2004. Carbon and nitrogen storage in agroforests, tree plantations, and pastures in western oregon, USA. *Agroforestry Systems* 60, 123-130.

Sheldrick, R., Auclair, D., 2000. Origins of agroforestry and recent history in the UK. In: Hislop, M., Claridge, J., (eds.) *Agroforestry in the UK*. Bulletin 122. Edinburgh: Forestry Commission.

Shennan, C., 2008. Biotic interactions, ecological knowledge and agriculture. *Philosophical Transaction of the Royal Society B* 363, 717-739.

Shepherd, M.A., Lord, E.I., 1996. Nitrate leaching from a sandy soil: the effect of previous crop and post-harvest soil management in an arable rotation. *Journal of Agricultural Science* 127(2), 215-229.

Shepherd, M. A., Harrison, R., Webb, J., 2002. Managing soil organic matter – Implications for soil structure on organic farms. *Soil Use and Management* 18**,** 284-292.

Shepherd, M., Pearce, B., Cormack, B., Philipps, L., Cuttle, S., Bhogal, A., Costigan, P., Unwin, R., 2003. *An assessment of the environmental impacts of organic farming*, A review for Defra project OF0405. London: Department for Environment, Food and Rural Affairs (with ADAS, EFRC and IGER).

Short, I.J., 2006. *Outputs, ecological interactions and bioeconomic modelling of a novel silvopastoral system in lowland Ireland.* PhD thesis. Belfast: Queen's University.

Sibbald, A., 2006. Silvopastoral agroforestry: a land use for the future. *Scottish Forestry* 60, 4-7.

Sinclair, F.L., Eason, W.R., Hooker, J., 2000. Understanding and management of interactions. In: Hislop, A.M., Claridge, J., (eds.) *Agroforestry in the UK*. Bulletin 122. Edinburgh: Forestry Commission.

Smaje, C., 2014. Kings and commoners: Agroecology meets consumer culture. *Journal of Consumer Cultu*re 14(3), 365-383.

SMI, 2005. *A guide to managing crop establishment.* Chester: The UK Soil Management Initiative Ltd.

Smil, V., 2000. *Feeding the World: a Challenge for the 21st Century.* Cambridge, MA: MIT Press

Smith, H.G., Dänhardt, J., Lindström, A., Rundlöf, M., 2010. Consequences of organic farming and landscape heterogeneity for species richness and abundance of farmland birds. *Oecologia* 162, 1071-1079.

Smith, J., (unpublished). Data collected as part of CO-FREE project trials. Newbury: Organic Research Centre.

Smith, J., Pearce, B., Wolfe, M.S., 2012. A European perspective for developing modern multifunctional agroforestry systems for sustainable intensification. *Renewable Agriculture and Food Systems* 27(4), 323 - 332.

Smith, J., Pearce, B.D., Wolfe, M.S., 2013a. Reconciling productivity with protection of the environment: Is temperate agroforestry the answer? *Renewable Agriculture and Food Systems* 28(1), 80-92.

Smith, J., Westaway, S., Pearce, B., Lampkin, N., Briggs, S., 2013b. *Can agroforestry deliver production and environmental benefits in the next rural development programme?* Report to Natural England. Newbury: Organic Research Centre.

Smith, L.G., Little, T., 2013. *Environmental footprinting for farm businesses.* Aberystwyth: Organic Centre Wales.

Smith, L.G, Padel, S., Pearce, B., 2011. *Soil carbon sequestration and organic farming: An overview of current evidence.* Aberystwyth: Organic Centre Wales.

Smith, L.G., Gerrard, C.L., Padel, S., Pearce, B., Lampkin, N., 2014a. *Testing the sustainability of organic crop yields and rotations.* Report to Defra on project OF03100. Newbury: Organic Research Centre.

Smith, L.G., Williams, A.G., Pearce, B.D., 2014b. The energy efficiency of organic agriculture: A review. *Renewable Agriculture and Food Systems* FirstView, 1-22.

Smith, R.F. Bosch, R.v.d., 1967. Integrated control. In: Kilglore, W.W., Doutt, R.L., (eds.) *Pest control: biological, physical and selected chemical methods*. New York: Academic Press, 295-340.

Snapp, S., Swinton, S., Labarta, R., Mutch, D., Black, J., Leep, R., Nyiraneza, J., O'Neil, K., 2005. Evaluating cover crops for benefits, costs and performance within cropping system niches. *Agronomy Journal* 97(1), 322-332.

Snyder, C., Spaner, D., 2010. The sustainability of organic grain production on the Canadian prairies—a review. *Sustainability* 2(4), 1016-1034.

Soil Association, 2008. *Soil Association Organic Standards.* Bristol: Soil Association.

Song L, Zhang Dw, Li Fm, Fan, X., Ma, Q., Turner, N., 2010. Drought stress: Soil water availability alters the inter- and intra-cultivar competition of three spring wheat cultivars bred in different eras. *Journal of Agronomy and Crop Science* 196(5), 323-335.

Soon, Y.K., Clayton, G.W., Rice, W.A., 2001. Tillage and previous crop effects on dynamics of nitrogen in a wheat–soil system. *Agronomy Journal* 93(4), 842-849.

Spiertz, J., 2010. Nitrogen, sustainable agriculture and food security. A review. *Agronomy for Sustainable Development* 30(1), 43-55.

Stamps, W.T., Linit, M.J., 1998. Plant diversity and arthropod communities: Implications for temperate agroforestry. *Agroforestry Systems* 39, 73-89.

Stanhill, G., 1990. The comparative productivity of organic agriculture. *Agricultural Ecosystems and Environment* 30, 1-26.

Steiner, R., 1924. *Agriculture.* London: The Biodynamic Farming and Gardening Association Inc.

Stern, V., Smith, R., Bosch, R.v.d., Hagen, K., 1959a. The integration control concept. *Hilgardia* 29, 81-101.

Stern, V.M., Smith, R.F., Bosch, R.v.d., Hagen, K.S., 1959b. *The integration of chemical and biological control of the spotted alfalfa aphid*. University of California.

Stinner, W., Möller, K., Leithold, G., 2008. Effects of biogas digestion of clover/grass-leys, cover crops and crop residues on nitrogen cycle and crop yield in organic stockless farming systems. *European Journal of Agronomy* 29, 125-134.

Stoate, C., Szczur, J., Aebische, N.J., 2003. Winter use of wild bird cover crops by passerines on farmland in northeast England: Declining farmland species were more abundant in these crops which can be matched to the birds' requirements. *Bird Study* 50(1), 15-21.

Stolba, A., Woodgush, D.G.M., 1989. The behaviour of pigs in a semi-natural environment. *Animal Production* 48(2), 419-425.

Stolze, M., Piorr, A., Häring, A., Dabbert, S., 2000. *The environmental impacts of organic farming in Europe.* Stuttgart-Hohenheim: University of Hohenheim.

Stopes, C., Lord, E.I., Philipps, L., Woodward, L., 2002. Nitrate leaching from organic farms and conventional farms following best practice. *Soil Use and Management* 18, 256-263.

Suneson, C.A., 1956. An evolutionary plant breeding method. *Agronomy Journal* 48, 188-191.

Suter, M., Connolly, J., Finn, J.A., Loges, R., Kirwan, L., Sebastià, M.T., Lüscher, A. 2015. Nitrogen yield advantage from grass–legume mixtures is robust over a wide range of legume proportions and environmental conditions. *Global Change Biology* 21(6):2424–2438.

Sylvester-Bradley, R., Kindred, D.R., Blake, J., Dyer, C.J., Sinclair, A.H., 2008. *Optimising fertiliser nitrogen for modern wheat and barley crops.* Report to HGCA (Project 3084). Boxworth, Cambridge: ADAS.

Tabashnik, B.E., Brévault, T., Carrière, Y., 2013. Insect resistance to Bt crops: lessons from the first billion acres. *Nature Biotechnology* 31, 510–521.

Taylor, B.R., Younie, D., Matheson, S., Coutts, M., Mayer, C., Watson, C.A., 2006. Output and sustainability of organic ley/arable crop rotations at two sites in northern Scotland. *Journal of Agricultural Science* 144, 435–447.

Teasdale, J.R., 1993. Interaction of light soil moisture and temperature with weed suppression by hairy vetch residue. *Weed Science* 1993, 46-51.

Teasdale, J.R., 1996. Contribution of cover crops to weed management in sustainable agricultural systems. *Journal of Production Agriculture* 9(4), 475-479.

Teasdale, J.R., Beste, C.E., Potts, W.E., 1991. Response of weeds to tillage and cover crop residue. *Weed Science* 39(2), 195-199.

Teklehaimanot, Z., Mmolotsi, R.M., 2007. Contribution of red alder to soil nitrogen input in a silvopastoral system. *Biology and Fertility of Soils* 43, 843-848.

Tew, T.E., Macdonald, D.W., Rands, M.R.W., 1992. Herbicide application affects microhabitat use by arable wood mice (*Apodemus sylvaticus*). *Journal of Applied Ecology* 29(2), 532-539.

Thevathasan, N.V., Gordon, A.M., 2004. Ecology of tree intercropping systems in the north temperate region: Experiences from Southern Ontario, Canada. *Agroforestry Systems* 61, 257-268.

Thomas, M., Wratten, S., Sotherton, N., 1991. Creation of 'island' habitats in farmland to manipulate populations of beneficial arthropods: predator densities and emigration. *Journal of Applied Ecology* 28, 906-917.

Thomas, M., Wratten, S., Sotherton, N., 1992. Creation of 'island' habitats in farmland to manipulate populations of beneficial arthropods: predator densities and species composition. *Journal of Applied Ecology* 29, 524-531.

Thomas, T., Willis, R., 2000. The economics of agroforestry in the UK. In: Hislop, M., Claridge, J., (eds.) *Agroforestry in the UK.* Bulletin 122. Edinburgh: Forestry Commission.

Thomassen, M.A., Calker, K.J.v., Smits, M.C.J., Iepema, G.L., Boer, I.J.M.d., 2008. Life cycle assessment of conventional and organic milk production in the Netherlands. *Agricultural Systems* 96(1–3), 95-107.

Tilman, D., Reich, P., Knops, J., 2006. Biodiversity and ecosystem stability in a decade-long grassland experiment. *Nature* 441, 629-632.

Titonell, P. (2014) Agroecological solutions for future farming. Presentation to ORC Organic Producers' Conference: *Practical research and innovation - diversity in practice*, Solihull, November 2014. Newbury: Organic Research Centre, www.organicresearchcentre.com.

Torstensson, G., Aronsson, H., Bergström, L., 2006. Nutrient use efficiencies and leaching of organic and conventional cropping systems in Sweden. *Agronomy Journal* 98(3), 603-615.

Tscharntke, T., Clough, Y., Wanger, T.C., Jackson, L., Motzke, I., Perfecto, I., Vandermeer, J., Whitbread, A., 2012, Global food security, biodiversity conservation and the future of agricultural intensification. *Biological Conservation* 151, 53–59.

Tuck, S.L., Winqvist, C., Mota, F., Ahnstrom, J., Turnbull, L.A., Bengtsson, J., 2014. Land-use intensity and the effects of organic farming on biodiversity: A hierarchical meta-analysis. *Journal of Applied Ecology* 51, 746-755.

Tuomisto, H.L., Hodge, I.D., Riordan, P., Macdonald, D.W., 2012. Does organic farming reduce environmental impacts? A meta-analysis of European research. *Journal of Enviornmental Management* 112, 309-320.

Udawatta, R.P., Anderson, S.H., Gantzer, C.J., Garrett, H.E., 2006. Agroforestry and grass buffer influence on macropore characteristics: A computed tomography analysis. *Soil Science Society of America Journal* 70, 1763-1773.

Udawatta, R.P., Gantzer, C.J., Anderson, S.H., Garrett, H.E., 2008a. Agroforestry and grass buffer effects on pore characteristics measured by high-resolution x-ray computed tomography. *Soil Science Society of America Journal* 72(2), 295-304.

Udawatta, R.P., Garrett, H.E., Kallenbach, R.L., 2010. Agroforestry and grass buffer effects on water quality in grazed pastures. *Agroforestry Systems* 79(1), 81-87.

Udawatta, R.P., Kremer, R.J., Adamson, B.W., Anderson, S.H., 2008b. Variations in soil aggregate stability and enzyme activities in a temperate agroforestry practice. *Applied Soil Ecology* 39, 153-160.

Udawatta, R.P., Krstansky, J.J., Henderson, G.S., Garrett, H.E., 2002. Agroforestry practices, runoff, and nutrient loss: A paired watershed comparison. *Journal of Environmental Quality* 31, 1214-1225.

Ulber, L., Steinmann, H.-H., Limek, S., Isselstein, J., 2009. An on-farm approach to investigate the impact of diversified crop rotations on weed species richness and composition in winter wheat. *Weed Research* 49, 534-543.

UNCTAD, 2013. *Wake up before it is too late. Make agriculture truly sustainable now for food security in a changing climate.* Geneva: United Nations Conference on Trade And Development.

UNEP, 2011. *Emerging issues in our global environment.* Yearbook. Nairobi: United Nations Environment Programme.

Vaarst, M., Nissen, T.B., Østergaard, S., Klaas, I.C., Bennedsgaard, T.W., Christensen, J., 2007. Danish stable schools for experiential common learning in groups of organic dairy farmers. *Journal Dairy Science,* 90, 2543-2554.

Vaderstad, SMI, 2004. *Target on establishment.* Grantham and Chester: Vaderstad and Soil Management Initiative.

Vaderstad, SMI, 2006. *Visual soil assessment*. Grantham and Chester Vaderstad and Soil Management Initiative.

Vanbergen, A.J., Initiative, T.I.P., 2013. Threats to an ecosystem service: pressures on pollinators. *Frontiers in Ecology and the Environment* 11, 251-259.

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

Vanloqueren, G., Baret, P.V., 2009. How agricultural research systems shape a technological regime that develops genetic engineering but locks out agroecological innovations. *Research policy* 38(6), 971-983.

Varah, A., Jones, H., Smith, J., Potts, S.G., 2013. Enhanced biodiversity and pollination in UK agroforestry systems. *Journal of the Science of Food and Agriculture* 93(9), 2073-2075.

Venkat, K., 2012. Comparison of twelve organic and conventional farming systems: A life cycle greenhouse gas emissions perspective. *Journal of Sustainable Agriculture* 36(6), 29.

Verhulst, N., Govaerts, B., Nelissen, V., Sayre, K.D., Crossa, J., Raes, D., Deckers, J., 2011. The effect of tillage, crop rotation and residue management on maize and wheat growth and development evaluated with an optical sensor. *Field Crops Research* 120(1), 58-67.

Vickery, J., Carter, N., Fuller, R.J., 2002. The potential value of managed cereal field margins as foraging habitats for farmland birds in the UK. *Agriculture Ecosystems and Environment* 89, 41-52.

Vieweger, A., Doring, T., 2015. Assessing health in agriculture - towards a common research framework for soils, plants, animals, humans and ecosystems. *Journal of the Science of Food and Agriculture* 95(3), 438-46.

Vieweger, A.; Haering, A.M.; Padel, S.; Doering, T.F.; Ekert, S.; Lampkin, N.H.; Murphy-Bokern, D; Otto, K., 2014. The evaluation of the German Programme for Organic Food and Farming Research: Results and pointers for the future. In: Rahmann, G., Aksoy, U., (eds.) *Building Organic Bridges* Vol 2*,* International Society of Organic Agriculture Research (ISOFAR), Report 20. Braunschweig: Thünen-Institut, 351-354.

Walker, B.H., 1992. Biodiversity and ecological redundancy. *Conservation Biology* 6, 18-23.

Wall, D.H., Bardgett, R.D., Behan-Pelletier, V., Herrick, J.E., Jones, T.H., Six, J., Strong, D.R., (eds.) 2012. *Soil Ecology and Ecosystem Services.* Oxford: Oxford University Press.

Warner, K. D., 2007, The quality of sustainability: Agroecological partnerships and the geographic branding of California wine grapes. *Journal of Rural Studies* 23(2), 142-155.

Watson, C., Atkinson, D., Gosling, P., Jackson, L., Rayns, F., 2002. Managing soil fertility in organic farming systems. *Soil Use and Management* 18, 239-247.

Watson, C.A., Baddeley, J.A., Edwards, A.C., Rees, R.M., Walker, R.L., Topp, C.F.E., 2011. Influence of ley duration on the yield and quality of the subsequent cereal crop (spring oats) in an organically managed long-term crop rotation experiment. *Organic Agriculture* 1, 147–159.

Watson, C.A., Chamberlain, D.E., Norton, L.R., Fuller, R.J., Atkinson, C.J., Fowler, S.M., McCracken, D.I., Wolfe, M.S., Walker, R.L., 2006. Can organic farming deliver natural heritage goals in the UK uplands. *Aspects of Applied Biology* 79, 5-8.

Weigelt, A., Weisser, W., Buchmann, N., Scherer-Lorenzen, M., 2009. Biodiversity for multifunctional grasslands: Equal productivity in high-diversity low-input and low-diversity high-input systems *Biogeosciences* 6, 1695-1706.

Weiner, J., Griepentrog, H., Kristensen, L., 2001. Suppression of weeds by spring wheat triticum aestivum increases with crop density and spatial uniformity. *Journal of Applied Ecology* 38(4), 784-790.

Weinert, T.L., Pan, W.L., Moneymaker, M.R., Santo, G.S., Stevens, R.G., 2002. Nitrogen recycling by nonleguminous winter cover crops to reduce leaching in potato rotations. *Agronomy Journal* 94(2), 365-372.

Weltzien, H.C., 1991. Biocontrol of foliar fungal diseases with compost extracts. In: *Microbial Ecology of Leaves.* Brock/Springer Series in Contemporary Bioscience, 430-450.

Werf, W.v.d., Keesman, K., Burgess, P.J., Graves, A.R., Pilbeam, D.J., Incoll, L.D., Metselaar, K., Stappers, R., Keulen, H.v., Palma, J., Dupraz, C., 2007. Yield-safe: A parameter-sparse process-based dynamic model for predicting resource capture, growth and production in agroforestry systems. *Ecological Engineering* 29, 419-433.

Wezel, A., Bellon, S., Dore, T., Francis, C., Vallod, D., David, C., 2009. Agroecology as a science, a movement and a practice. A review. *Agronomy for Sustainable Development* 29, 503-515.

White, P.C.L., Hassall, M., 1994. Effects of management on spider communities of headlands in cereal fields. *Pedobiologia* 38(2), 169-184.

Wibbelmann, M.; Schmutz, U.; Wright, J.; Udall, D.; Rayns, F.; Kneafsey, M.; Trenchard, L.; Bennett, J., Lennartsson, M., 2013. *Mainstreaming Agroecology: implications for global food and farming systems.* Discussion paper. Coventry: University Centre for Agroecology and Food Security.

Wijnands, F.G., 1997. Integrated crop protection and environment exposure to pesticides: Methods to reduce use and impact of pesticides in arable farming. *Developments in Crop Science* 25, 319-328.

Willey, R.W., 1979. Intercropping - its importance and research needs. Part 1: Competition and yield advantages. *Field Crops Abstracts* 32, 1-10.

Williams, A.G., Audsley, E., Sandars, D.L., 2006. *Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities*. Main report to Defra IS0205. London: Department for Food, Environment and Rural Affairs.

Williams, B., Warren, J. 2004. Effects of spatial distribution on the decomposition of sheep faeces in different vegetation types. *Agriculture, Ecosystems and Environment* 103, 237–243.

Williams, P.A., Gordon, A.M., Garrett, H.E., Buck, L., 1997. Agroforestry in north America and its role in farming systems. In: Gordon, A.M., Newman, S.M., (eds.) *Temperate Agroforestry Systems*. Wallingford: CABI Publishing, 9-84.

Williams, P.A., Koblents, H., Gordon, A.M., 1995. Bird use of an intercropped maize and old fields in Southern Ontario. In: Ehrenreich, J.H., Ehrenreich, D.L., (eds.), *Proceedings of the Fourth North American Agroforestry Conference 1995*. Boise, Idaho, United States, 158-162.

Wink, M., 2012. Medicinal plants: A source of anti-parasitic secondary metabolites. *Molecules,* 17(11), 12771-12791.

Winqvist, C., Ahnstrom, J., Bengtsson, J., 2012. Effects of organic farming on biodiversity and ecosystem services: Taking landscape complexity into account. *Annals of the New York Academy of Sciences* 1249, 191-203.

Wolfe, M.S., 1985. The current status and prospects of multiline cultivars and variety mixtures for disease resistance. *Annual Review of Phytopathology* 23, 251-273.

Wolfe, M.S., 1992. Barley diseases: Maintaining the value of our varieties. In: Munck, L., Kirkegaard, K., Jensen, B., (eds.) *Barley genetics VI*. Proceedings of the 6th International Barley Genetics Symposium. Vol. II, Helsingborg, Sweden, 1055-1067.

Wolfe, M.S., Baresel, J.P., Desclaux, D., Goldringer, I., Hoad, S., Kovacs, G., Löschenberger, F., Miedaner, T., Østergård, H., Lammerts v. Bueren, E.T., 2008. Developments in breeding cereals for organic agriculture. *Euphytica* 163(3), 323-346.

Wolfenbarger, L.L., Phifer, P.R., 2000. The ecological risks and benefits of genetically engineered plants. *Science* 290(5499), 2088-2093.

Woodland Trust, 2013. *The Pontbren project: A farmer-led approach to sustainable land management in the uplands.* Grantham: Woodland Trust.

Worthington, M., Reberg-Horton, C., 2013. Breeding cereal crops for enhanced weed suppression: Optimizing allelopathy and competitive ability. *Journal of Chemical Ecology* 39(2), 213-231.

Wrage, N., Strodthoff, J., Cuchillo-Hilario, M., Isselstein, J., Kayser, M., 2011. Phytodiversity of temperate permanent grasslands: Ecosystem services for agriculture and livestock management for diversity conservation. *Biodiversity and Conservation* 20, 3317-3339.

Wratten, S.D., Gillespie, M., Decourtye, A., Mader, E., Desneux, N., 2012. Pollinator habitat enhancement: Benefits to other ecosystem services. *Agriculture, Ecosystems and Environment* 159, 112-122.

Wright, C., 1994. The distribution and abundance of small mammals in a silvoarable agroforestry system. *Agroforestry Forum* 5(2), 26-28.

Wu, H., Pratley, J., Lemerle, D., Haig, T., 1999. Crop cultivars with allelopathic capability. *Weed Research* 39(3), 171-180.

Wyland, L.J., Jackson, L.E., Chaney, W.E., Klonsky, K., Koike, S.T., Kimple, B., 1996. Winter cover crops in a vegetable cropping system: Impacts on nitrate leaching, soil water, crop yield, pests and management costs. *Agriculture, Ecosystems and Environment* 59(1–2), 1-17.

Yates, C., Dorward, P., Hemery, G., Cook, P., 2007. The economic viability and potential of a novel poultry agroforestry system. *Agroforestry Systems* 69, 13-28.

Yobterik, A.C., Timmer, V.R., Gordon, A.M., 1994. Screening agroforestry tree mulches for corn growth: A combined soil test, pot trial and plant analysis approach. *Agroforestry Systems* 25, 153-166.

Young, A., 1997. *Agroforestry for Soil Management.* Wallingford: CABI Publishing.

Young, J., Griffin, M., Alford, D., Ogilvy, S., (eds.) 2001. *Reducing agrochemical use on the arable farm: The Talisman and Scarab projects.* London: Department for Environment, Food and Rural Affairs.

Younie, D., 2012. *Grassland management for organic farmers.* Marlborough: The Crowood Press.

Younie, D., Watson, C.A., 1992. Soil nitrate-N levels in organically and intensively managed grassland systems. *Aspects of Applied Biology* 30, 235-238.

Zehetmeier, M., Baudracco, J., Hoffmann, H., Heißenhuber, A., 2012. Does increasing milk yield per cow reduce greenhouse gas emissions? A system approach. *Animal* 6(1), 154-166.

Zentner, R.P., Lafond, G.P., Derksen, D.A., Nagy, C.N., Wall, D.D., May, W.E., 2004. Effects of tillage method and crop rotation on non-renewable energy use efficiency for a thin black chernozem in the Canadian prairies. *Soil and Tillage Research* 77(2), 125-136.

Zhu, Y., Chen, H., Fan, J., Wang, Y., Li, Y., Chen, J., Fan, J.X., Yang, S., Hu, L., Leung, H., Mew, T.W., Teng, P.S., Wang, Z., Mundt, C.C., 2000. Genetic diversity and disease control in rice. *Nature* 406, 718-722.