CRANFIELD UNIVERSITY

Laurence Smith

The impacts of a large-scale conversion to organic agriculture in England and Wales

School of Water, Energy & Environment
PhD in Agricultural Systems Modelling

Academic Year: 2016 - 2017

Supervisors: Dr Adrian Williams, Prof. Guy Kirk, Dr Bruce Pearce

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This thesis is submitted in partial fulfilment of the requirements for the degree of PhD

ABSTRACT

With the need to identify sustainable modes of food production for growing populations there has been a growing interest in the potential of organic farming. Although evidence suggests that organic systems can produce food in an environmentally efficient manner, the impacts of a widespread conversion to organic management are still uncertain. The research presented aimed to address this knowledge gap by completing a comprehensive and robust assessment of the food production, fossil energy-use and greenhouse gas impacts associated with a 100% conversion to organic farming in England and Wales.

Firstly a structured literature review was carried out to determine the relative fossil-energy efficiency of organic systems. The sustainability of typical organic crop rotations was then assessed using a simulation model of crop-soil N dynamics. Land-use and production scenarios under 100% organic management were assessed through the development and application of a large-scale linear programming model that estimates levels of production under biophysical constraints, e.g. N supplies from biological fixation by legumes. A life-cycle assessment-based model was then applied to explore the extent to which a 100% conversion to organic farming could lead to improvements in greenhouse gas mitigation and fossil energy efficiency. The environmental assessment approach allowed for processes inside and outside of the immediate boundaries of the production systems to be assessed, with the question “what is affected by the change in levels of production?” asked throughout the process.

The results revealed that whilst some organic systems offer improved performance in non-renewable resource use efficiency, a widespread conversion would result in a substantial decrease in domestic food production. Total food output expressed over five major food groups fell to 64% of a non-organic baseline. An increase in food imports would therefore be required to meet demand. From a greenhouse gas perspective, a 100% conversion to organic farming in England and Wales could lead to 6% decrease in the impacts

of food production. The greenhouse gas mitigation potential of organic farming is strongly related to the use of clover and other legumes in place of manufactured N and lower concentrate feed rates in livestock production. Where the additional-land required under an organic scenario is newly cultivated, it is likely that any greenhouse gas benefit obtained would be offset. Total greenhouse gas emissions increased by an average of 28%, compared to a non-organic baseline, when the land use change impacts associated with increased food imports were included. When the soil carbon sequestration benefits obtained through organic farming are also included the net difference between the two production systems is lessened, however a fundamental question remains concerning the availability of overseas land (land use requirements under organic management increased by 29-47% depending on the scenario).

Reducing the area of fertility-building ley within organic rotations is likely to improve productivities and reduce land-use requirements within organic farming systems. Improving crop cultivation practices, more effective cover-cropping and improved biological N-fixation could also help to improve N efficiency and productivity within organic systems. Changes to international organic standards in some areas may also improve the environmental sustainability of the sector, e.g. by allowing recycling of P from sewage treatment.

Overall the research showed that whilst the adoption of organic farming can lead to improvements in environmental performance, a widespread conversion would need to be accompanied by substantial changes in diet and/or typical organic practices to become feasible from the perspectives of environmental impact and total food production.

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ACKNOWLEDGEMENTS

I am extremely grateful to Adrian Williams, Guy Kirk and Bruce Pearce, academic supervisors at Cranfield University and the Organic Research Centre (ORC), for their constructive comments, advice, guidance and support. Particular thanks are also due to Philip Jones at the Centre for Agricultural Strategy, Reading University, for his assistance with the application of the GAMS programming language and for his continual support. I am also grateful to Richard Tranter for allowing me to work at the Centre for Agricultural Strategy for a three-month period in 2014. Thank-you also to Daniel Sandars, Eric Audsley and Kerry Pearn at Cranfield and Exeter Universities for their assistance and advice on the application of linear programming models.

I am very grateful to the agricultural systems modellers at Scotland's Rural College (SRUC), in particular Kairsty Topp, Davide Tarsitano and Stephanie Jones for their assistance with the nutrient management in organic crop rotations study. Particular thanks are also due to Geert-Jan Van der Burgt for his technical guidance on the adaptation and application of the NDICEA model.

Furthermore, I wish to thank all my colleagues at the ORC for a positive and helpful working environment and the Centre's Director, Nic Lampkin, for allowing me to take study-leave in 2015-2016. Particular thanks are also due to Catherine Gerrard, whom I worked alongside over a five-year period at ORC, and whose support and statistical advice were invaluable.

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Laurence Smith

School of Water, Energy & Environment, Cranfield University and The Organic Research Centre, July 2017

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LIST OF ABBREVIATIONS

AHDB	Agriculture and Horticulture Development Board
BPEX	British Pig Executive
CAPRI	Common Agricultural Policy Regionalised Impact Analysis
CFE	Combined Food and Energy
EEC	European Economic Community
EU	European Union
FADN	Farm Accountancy Data Network
FAO	Food and Agriculture Organisation
FARMIS	FARM Information System
FBS	Farm Business Survey
GAMS	General Algebraic Modelling System
GHG	Greenhouse Gases
GIS	Geographic Information System
IFOAM	International Federation of Organic Agriculture Movements
IMPACT	International Model for Policy Analysis of Agricultural Commodities and Trade
IPCC	Intergovernmental Panel on Climate Change
JAC	June Agricultural Census
LCA	Life Cycle Assessment / Life Cycle Analysis
LP	Linear Programming
LU	Livestock Unit
LUAM	Land Use Allocation Model
LUC	Land Use Change
NDICEA	Nitrogen Dynamics In Crop rotations in Ecological Agriculture
OLUM	Optimal Land Use Model
RMSE	Root Mean Square Error
SOC	Soil Organic Carbon
SOM	Soil Organic Matter
UNFCCC	United Nations Framework Convention on Climate Change

CHAPTER 1. INTRODUCTION

This chapter provides context for the work carried out, before outlining the key hypothesis and the contribution to knowledge. An overview of the thesis layout and the interaction between chapters is also provided in addition to a disclosure statement that describes the contribution of co-authors within chapters 4-7.

1.1 Background

The extent to which contrasting systems of food production can contribute to increasing demands is a topic of much debate. At the same time changing weather patterns and limited availability of natural resources are posing increasing challenges to fragile ecosystems. Increased wealth and urbanisation are also changing the diet of global populations towards animal protein consumption and agriculture's impact on the environment is set to increase (Lampkin *et al.*, 2015). The magnitude of these issues has led to calls for radical shifts in the way food is produced, and moves towards forms of agriculture that can produce more food from less resource whilst sustaining rural livelihoods and public health (McIntyre *et al.*, 2008; Foresight, 2011).

Organic agriculture seeks to address many of the public demands for high quality, safe food, produced with minimal environmental losses (de Boer, 2003). However the differences between the environmental burdens of organic and conventional farming systems are still uncertain. Conventional farmers are also adopting 'organic practices' such as cover cropping, use of organic fertilisers and diverse rotations to improve resource use efficiencies and reduce pest, disease and weed burdens (Wezel *et al.* 2014) making the differences between the two systems less distinct. In addition, whilst organic farming provides some benefits over conventional systems, through the inclusion of practices that can reduce reliance on manufactured inputs, it has been suggested that there are some considerable disadvantages to the organic approach (Trewavas, 2001). For example, the need for intensive soil cultivation to manage weeds, and the increased nitrous oxide emissions associated with incorporation of green manure and cover crops (Olesen, 2009). Moreover, whilst the environmental impacts per unit of area are often less under organic agriculture (Aubert *et al.*, 2009), when comparisons are made on a product basis, organic systems can perform worse, primarily as a result of lower yields, resulting from limited N availability and worse feed conversion in some livestock sectors (Williams *et al.*, 2006; Seufert *et al.*, 2012).

Given the lower productivity of organic systems, it is important to know the consequences of a major shift to organic agriculture for national and global food production, without major changes in consumer diets and/or an increase in agricultural land area (Foresight, 2011). How much of the recorded environmental benefit from organic agriculture would be offset through a higher demand for land due to lower yields is questioned (Leifeld and Fuhrer, 2010) and the extent to which a large-scale uptake of organic practices would influence agriculture's contribution to global warming is still uncertain. Although some attempts have been made to tackle these questions (Audsley *et al.*, 2009; Jones and Crane, 2009), previous upscaling studies have been limited in their scope. A further question concerns the extent to which a widespread conversion to organic farming would support calls for healthier diets, especially lower meat and increased fresh vegetable consumption (Buttriss, 2016).

1.2 Specification of the working hypotheses for this study

The work presented in this thesis aimed to assess the consequences of a widespread shift to organic farming, and the benefits and/or dis-benefits of the organic approach. The overarching research question was 'what are the production and environmental impacts of a large-scale conversion to organic agriculture within England and Wales?'. The key hypotheses for the model-based analysis were defined as follows:

Hypothesis 1:

A 100% conversion of agriculture in England and Wales to organic practices will not significantly reduce the levels of production of major arable and horticultural crops and livestock products.

Hypothesis 2:

A 100% conversion of agriculture in England and Wales to organic practices will not result in a net increase in greenhouse gas emissions.

Hypothesis 3:

A 100% conversion of agriculture in England and Wales to organic practices will result in less fossil-fuel use per kg of product

Hypothesis 4:

A 100% conversion of agriculture in England and Wales to organic practices will result in less fossil-fuel use per hectare of land.

1.3 Research contribution

The study built on previous work (e.g. Lampkin, 1994; Jones and Crane, 2009) by developing a more representative and detailed modelling framework that better reflects typical organic farm structures and organic yields under conditions of limited N supply. The specific novel contributions to knowledge and understanding are as follows:

- Firstly a comprehensive literature review on the energy efficiency of organic agriculture adds further information on the extent to which organic practices can contribute to greater fossil fuel efficiency, and the extent of any differentiation between farming systems with a focus on systems operating in a European, US, Canadian and Australasian context
- Secondly a dynamic approach to nitrogen modelling explores the sustainability of contrasting organic crop rotations from a nitrogen, phosphorus and potassium supply and demand perspective. The same

modelling approach is applied to explore the extent to which soil types and rainfall affect N availability and crop productivity within organic cropping systems

- Thirdly a novel approach to scaling up organic agriculture explores the extent to which a widespread uptake of organic farming could meet current levels of production for key commodities. This modelling approach builds on previous work by ensuring that key 'limiting factors' are accounted for.
- Finally a Life Cycle Assessment (LCA) based model explores the extent to which a widespread conversion to organic management could foster progress towards improved energy efficiency and greenhouse gas mitigation in agriculture in England and Wales, under the assumption that national diets remain the same.

1.4 Thesis layout

The research is presented in "paper" format and comprises four distinct peer-reviewed journal-standard articles and a book chapter (Table 1.1). A literature review is also presented in Chapter 3 to explore the common characteristics of organic systems in England and Wales, the growth and current status of the sector, the outputs from previous studies exploring the impacts of a 100% conversion and potential modelling approaches. An overall discussion and areas for future work are presented in Chapter 8 before overarching conclusions are drawn. The contribution of co-authors is summarised in an introductory summary page for each publication.

Table 1.1. Journal articles and book chapter presented in the thesis. Status: P = published, S = submitted, RS = Ready to submit.

Journal / book title	**Title of article / chapter**	**Status**	**Location**
Renewable Agriculture and Food Systems	The energy efficiency of organic agriculture: a review	P	Chapter 4
Improving organic animal farming	Can conversion to organic methods contribute to GHG mitigation and improved energy efficiency in livestock production?	S	Appendix A
Renewable Agriculture and Food Systems	Predicting the effect of rotation design on N, P, K balances on organic farms using the NDICEA model	P	Chapter 5 and Appendix B
Land Use Policy	Modelling the impact of a widespread conversion to organic agriculture in England and Wales	S	Chapter 6 and Appendices B,C,D
Global Change Biology	Modelling the greenhouse gas implications of conversion of food production in England and Wales to organic methods	RS	Chapter 7 and Appendix E

Paper 1 / Chapter 4 explores the fossil-energy efficiency of contrasting organic production systems, allowing for an identification of systemic differences between the two farming systems in terms of on-farm energy use (e.g. diesel, electricity) and off-farm or indirect energy (e.g. fossil fuel used in the production of feed or fertiliser).

Paper 2 / Chapter 5 assesses the performance of a dynamic model of soil nitrogen – NDICEA (Van der Burgt *et al.*, 2006) utilising data from long-term organic trials. In a second stage of work, the sustainability of a range of organic crop rotations is assessed from a nutrient management perspective using a range of soil types and rainfall conditions within the same modelling framework.

Paper 3 / Chapter 6 investigates the production impacts of scaling-up organic farming in England and Wales using a large-scale linear programming model and crop yields adjusted by soil and rainfall class. The model incorporates constraints relating to crop rotation, N availability, stocking rate, livestock feed composition and feed availability at a national scale. The results provide an updated and comprehensive overview of the impacts of scaling-up organic practices and identify some interventions which could improve or worsen performance of a widespread conversion through a sensitivity analysis. A detailed description of the model is contained in Appendix C.

Paper 4 / Chapter 7 provides an overview of the greenhouse gas and fossil energy impacts of a 'realistic' 100% organic scenario through an application of environmental Life Cycle Assessment (LCA). The results reveal the extent to which organic farming methods could reduce greenhouse gas emissions from agricultural systems in the UK, and the potential knock-on impacts of the reduced yields and increased imports in terms of land-use change overseas.

An overview of how each chapter relates to the hypothesis described in section 1.2 is shown in Figure 1.1. Linkages between the various chapters are also indicated:

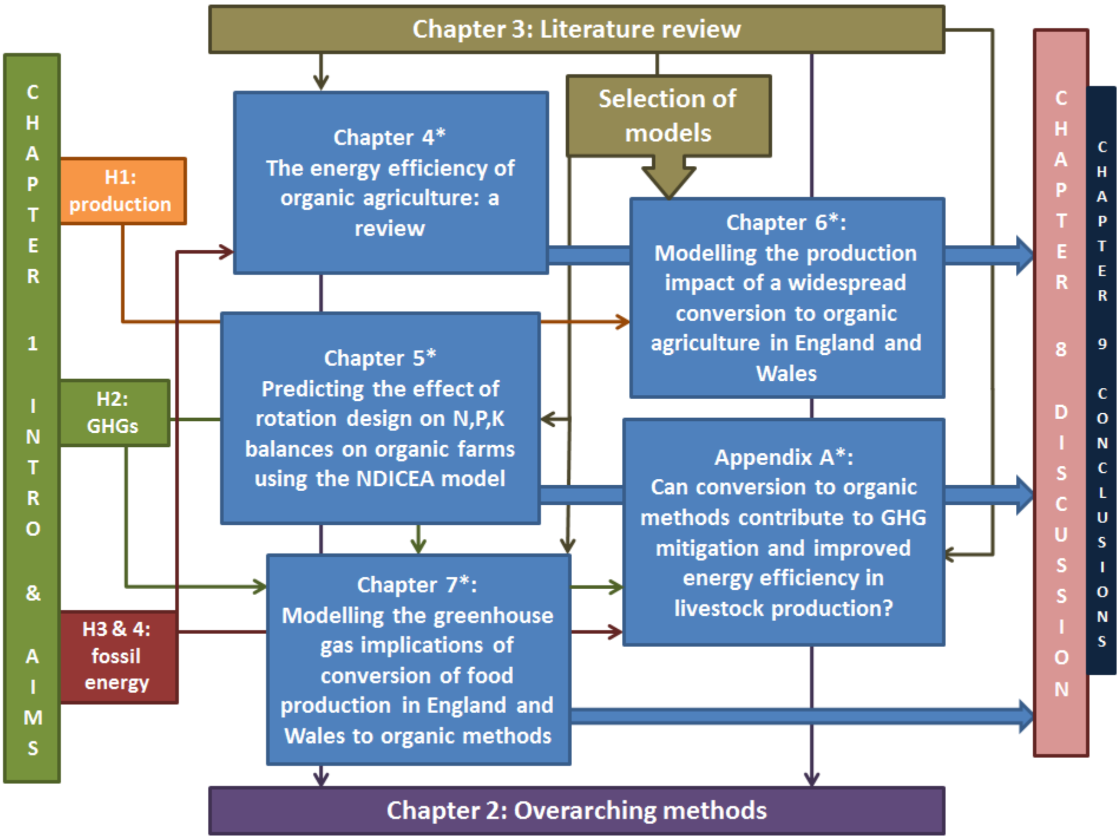

Figure 1.1. Overview of thesis chapters and their interactions. Asterisk indicates peer reviewed outputs. 'H1, H2, H3, H4' refers to Hypothesis 1, 2, 3 and 4 described in section 1.2.

1.5 Dissemination from PhD thesis

In addition to the peer-reviewed outputs described above, several conferences were attended over the course of the PhD to present early results from the analysis and obtain feedback on the methods applied. The following conference-papers and poster presentations were made over the lifetime of the thesis:

1.5.1 Conference papers

Smith, L.G., Goglio, P., Williams, A.G., 2016. Energy efficiency in organic farming systems. LCA Food Conference 2016, UCD, Dublin, 19-21st October 2016.

Smith, L.G., Tarsitano, D., Topp, C.F.E., Jones, S.K., Gerrard, C.L., Pearce, B.D., Williams, A.G., 2015. Assessing the influence of rotation design on the N, P, K balance of organic cropping systems. Aspects of Applied Biology: Valuing long-term sites and experiments for agriculture and ecology 128: 157-164. University of Newcastle, 27-28th May 2015.

Smith, L.G., Williams, A.G., Pearce, B.D., 2013. Energy use in organic farming. Tyndall Centre's Climate Transitions Conference, Cardiff University 4-5th April 2013.

1.5.2 Research posters

Smith, L.G., Jones, P.J., Pearce, B.D., Williams, A.G., 2017. Assessing the production impacts of a large-scale conversion to organic farming in England and Wales. In: Rahmann, G (ed.) Proceedings of the 19th Organic World Congress, New Dehli, India, November 9-11, 2017. Organized by ISOFAR/OFAI/TIPI.

Smith, L.G., Pearce, B.D., Williams, A.G., Kirk, G., 2017. Assessing the productivity of organic rotations using the NDICEA model. In: Rahmann, G (ed.) Proceedings of the 19th Organic World Congress, New Delhi, India, November 9-11, 2017. Organized by ISOFAR/OFAI/TIPI.

Smith, L.G., Tarsitano D, Topp C.F.E., Jones S.K., Gerrard C.L, Pearce B.D, Williams, A.G., Kirk G, Watson, C., 2015. Assessing the Nitrogen Balance of Organic farms using the NDICEA model. In: Baggs, E (ed.) Proceedings of the BSSS 2015 Annual Meeting: Celebrating a New Era for Soil Science, 26th November 2015.

1.6 Disclosure statement

Chapters 4, 5, 6, 7 are presented as co-authored papers. Contributions from the lead author (Laurence Smith) and the co-authors listed on each paper are outlined below:

Chapter 4: The literature search strategy, data collection and analysis were developed and implemented by the lead author. Adrian Williams at Cranfield

University and Bruce Pearce at the Organic Research Centre provided suggestions on relevant sources of literature and commented on the search strategy.

Chapter 5: The lead author planned the research approach and assessed the accuracy of the NDICEA model using data collected from long-term organic trials in England and Wales. The lead author also gathered data on typical organic farm structures through expert interviews and an assessment of Farm Business Survey data. The lead author applied the data collected to determine the sustainability of a range of organic crop rotations. Researchers at Scotland's Rural College (SRUC) assessed the accuracy of NDICEA at the Scottish sites and the nutrient balance of the organic crop rotations applied at SRUC.

Chapter 6: The lead author designed the research approach and developed a bespoke model, the Optimal Land Use Model (OLUM), for use within the study. As part of this process the lead author collected data on typical organic practices and produced model-derived organic crop yields for a range of typical organic crop rotations. Philip Jones (Reading University) provided regular input to the development of the OLUM whilst Adrian Williams and Guy Kirk (Cranfield University) provided input on the modelled scenarios selected for inclusion.

Chapter 7: The lead author designed the research approach and collected data from a range of industry sources in-order to adapt the Agri-LCA models. The lead author also accessed import/export data and results from overseas LCA-based studies to determine the environmental impacts of imported food. The lead author also collected data on carbon sequestration in organic farming systems and published estimates of the greenhouse gas impacts from land-use change. Adrian Williams provided an overview of the Agri-LCA models and access to relevant data for the calculation of the environmental impacts. Guy Kirk provided guidance on the calculation of the greenhouse gas emission offset from soil carbon sequestration in organic systems.

CHAPTER 2. OVERVIEW OF THE METHODS APPLIED IN THESIS

In order to answer the research questions posed in section 1.2 a range of approaches were applied throughout the various chapters. An overview of the methods applied in the PhD thesis is provided in the following section(s).

2.1 Structured literature reviews

The literature reviews aimed to identify previous work, provide a basis for the selection a suitable modelling framework (Chapter 3) and examine the extent to which various organic farming systems can improve production efficiencies with regard to fossil energy-use and greenhouse gas emissions (Chapters 3, 4 and Appendix A). The reviews were carried out with a range of web-based search engines (ISI Web of Knowledge, Scopus, Google Scholar, BIOSIS Previews, SCIRUS, Science Direct, Organic Eprints). Some non-certified systems were included in the comparisons, although these were required to adhere to the IFOAM (International Federation of Organic Agriculture) principles (see section 3.1.1). Grey literature were also included within the searches (i.e. PhD theses, Government and NGO reports and research project reports).

Within Chapter 3 a multi-criteria assessment was completed to compare the suitability of a range of models for use within this study. Criteria for this evaluation were selected through a discussion with PhD supervisors at Cranfield University and through expert input from the School of Agriculture, Policy and Development at Reading University (Mr. Philip Jones). A range of farm-level case-study portraits were included in the literature review presented in Appendix A, with the information on each case gathered through structured interviews. These interviews were carried out over the telephone or in person with each farmer / land-manager.

2.2 Nitrogen modelling

Chapter 5 aimed to assess the extent to which typical yields within organic crop rotations could be sustained under a range of environmental conditions, recognising that the supply of N can be a limiting factor within organic production systems (Berry *et al.*, 2002). As a preliminary step, the suitability of three models of soil N was assessed, i.e. the SUNDIAL model, developed at Rothamsted Research (Smith *et al.*, 1996), the EU-Rotate N model developed at Warwick University with assistance from partner institutes in the EU Quality of Life Programme (Rahn *et al.*, 2007) and the Nitrogen Dynamics In Crop rotations in Ecological Agriculture (NDICEA) model developed at Wageningen University and the Louis Bolk Institute in the Netherlands. Support for the use of SUNDIAL was found to be unavailable, whereas EU-Rotate N was unable capture the growth of grass/clover leys in organic rotations (although provision is made for this within the code, growth of the ley period was not captured in the modelled outputs, despite numerous attempts at debugging with the assistance of the model's developers). The NDICEA model was therefore selected and its accuracy assessed through a validation exercise.

2.2.1 Validation of the NDICEA model

Validation runs were completed through a comparison of measured and simulated values at a range of sites. Six suitable trial-sites were located in the UK, i.e. the Defra funded organic conversion trials at ADAS Terrington, HRI Huntsmill and Ty Gywn (IBERS), the Organic Research Centre's Stockless Trial and two long-term grassland/arable trials at SRUC. Figure 5.1 provides an overview of the location and site parameters for each trial. The data collected for the validation assessment is summarised in Table 2.1.

Table 2.1: soil, environment and cropping data extracted from research archives for a range of organic trials. The data collected were used to assess the accuracy of the NDICEA model

Soil data	Environment data	Cropping data
Soil organic matter (SOM) contents	Rainfall (mm)	Crop type
Soil N content (kg N ha^{-1})	Temperature (degrees C)	Crop yield (t DM ha^{-1} yr^{-1})
Topsoil depth (cm)	Evapotranspiration (mm)	Crop dry matter (%)
Soil type (clay, loam, sand, silt etc)	Atmospheric deposition data (Kg N ha^{-1} yr^{-1})	Cultivation dates (sowing, harvest dates etc)
pH		Fertilisation data (type and amount)
Maximum rooting depths (cm - if available)		Seed rates (kg ha^{-1})

Daily temperature, rainfall and evapotranspiration data were entered into the model through the use of Comma Separated Value (CSV) files. The other data (e.g. soil pH, crop dry matter percentages) were entered manually via the model's interface. Observed N and SOM values were used to apply an automatic calibration feature that adjusts default values controlling rates of N fixation, decomposition, N leaching and denitrification within NDICEA (Swain *et al.*, 2015). Measurements of soil N were then used to assess the model's accuracy (see Figure 2.1) and to calculate Root Mean Square Errors (RMSE), based on the difference between measured and simulated soil N values at each trial site (equation 1).

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(sim-obs)^2}{n}} \qquad (1)$$

In equation 1 *sim* refers to the predicted soil N content within NDICEA on a given day in a calendar year, *obs* refers to the recorded soil N content on the same day at a given site and *n* represents the total number of soil N samples over the time period assessed.

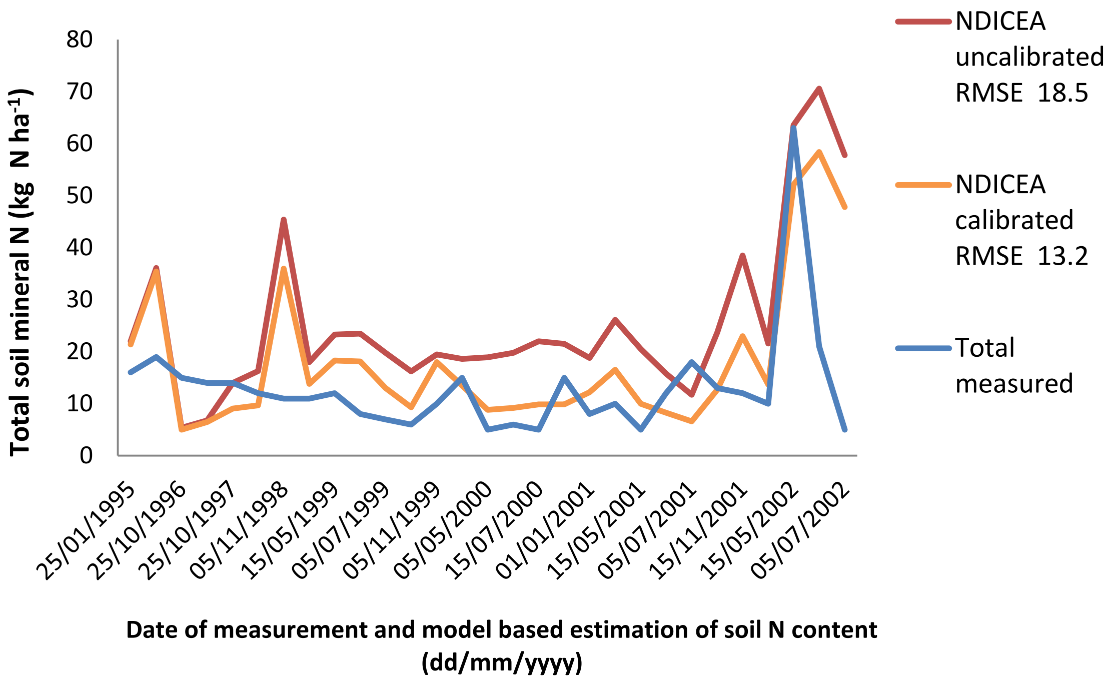

Figure 2.1. Comparison of the measured and modelled soil N content at SRUC Woodside plot W3711 on various dates from 1995 - 2002

The RMSE can be used to determine the residual variance, and hence the accuracy of models, under a range of conditions. A degree of transparency is also provided by reporting average prediction error in the same unit as the observed variable (in this case kg N ha^{-1}). For the purpose of this study, a RMSE of less than 20 kg N ha^{-1} yr^{-1} was deemed to represent good-enough performance for practical purposes (Van der Burgt *et al.*, 2006). As this could be achieved for most of the experimental sites (see Table 5.3) the model was used to assess the nutrient demand and supply of each long-term trial (the results from this work are illustrated in Chapter 5) .

2.2.2 Modelling organic yield potentials

Following the validation exercise, the NDICEA model was applied to determine N availabilities and potential crop yields within a wide range of organic rotations and for various soil types and rainfall zones in England and Wales. As the model is target-oriented (i.e. NDICEA 'reconstructs' the dynamics of the state variables required to achieve a yield entered by the user, Swain *et al.* 2015) average organic yield data were adjusted manually in-line with nitrogen availability over the course of a rotation, as illustrated in Figure 2.2. The nitrogen balance (kg N ha^{-1} yr^{-1}) of each rotation was also used to determine whether yields should be increased or decreased (i.e. if the total N balance was low or negative, yields were reduced and vice-versa in situations of increased supply).

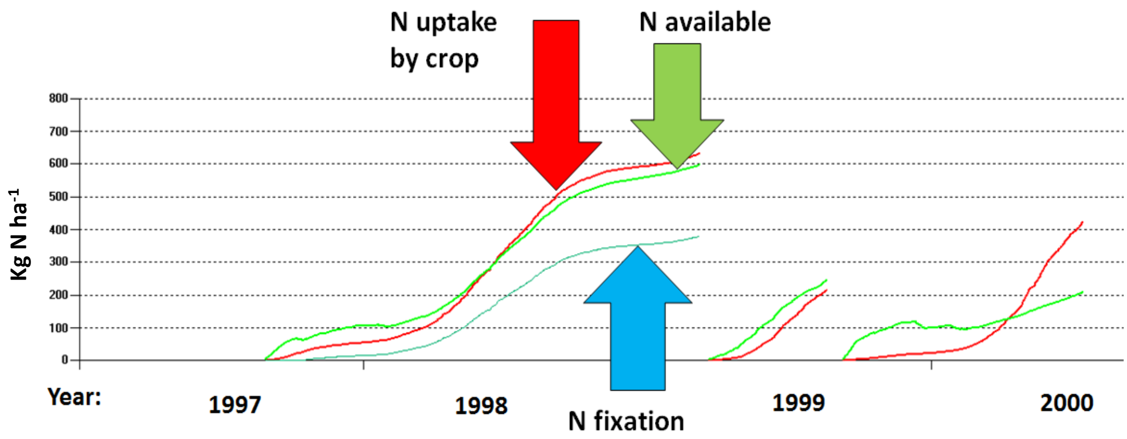

Figure 2.2. N availability predictions and crop uptake values reported within NDICEA interface. Average organic yields were increased / decreased in line with N availability over the course of the rotation

Table 2.2: Typical organic crop rotations assessed within NDICEA. Stocked rotations = with livestock manure; Stockless = without livestock manure

	Rotation year									
Rotation	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Stocked 'complex'	G/WC	G/WC	G/WC	WW	WO	RC/G	RC/G	P	SB	SW
Stocked 'simple'	RC/G	RC/G	WW	P	WW	WR				
Stockless 'complex'	RC/G	RC/G	P	WO	SB	SW				
Stockless 'simple'	RC/G	WW	PE	SO						
Field vegetable	RC/G	RC/G	P	BR	L					
Market garden	RC/G	RC/G	CB	O	B	C	SB	BR	PE	CG
Dairy	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	FB	WS	SB
Cattle and sheep	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	FB	WW
Mixed	G/WC	G/WC	G/WC	RC/G	WW	WO	SB	WB	WR	

(G/WC = Grass/white clover, WS = wholecrop silage, WB = winter barley, WW = winter wheat, WO = Winter oats, RC/G = red clover SW = spring wheat, SB = Spring beans, P = potatoes, WR = Winter rye, FB = Fodder beet, PE = peas, SO = spring oats, PE = peas BR = broccoli, L = leeks, CB = cabbage, O = onions, B = beetroot, C = carrots, CG = courgettes, SB = spring barley)

A range of typical organic rotations were assessed, using example cropping sequences within the Organic Farm Management Handbook (Lampkin *et al.*, 2012) and guidelines given to organic farmers on the recommended duration of fertility building leys in arable rotations (see Table 2.2, Lampkin, 2002; Lampkin *et al.*, 2012). Potential yields within each rotation were assessed for 16 soil/rainfall classes (see Table 2.3).

Table 2.3: Soil/rainfall classes used within the NDICEA based yield adjustment. A total of 16 (i.e. 4x4) classes were used for the yield adjustment exercise. 'Humose' refers to soil types with a low pH and high organic matter content

Soil classes	Rainfall classes
Light	Dry
Medium	Average
Heavy	Wet
Humose	Very Wet

Soil types within NDICEA (e.g. clay, loam, sand etc) were allocated to the above soil classes using a soil texture index from the Silsoe Whole Farm Model (Audsley *et al.*, 2006). Rainfall classes were derived from a 30-year UK Met Office dataset. Average organic crop yield data (i.e. pre-adjusted yields) and maximum/minimum yield ranges were obtained from structured interviews with IOTA[1] registered advisors, sources providing technical information to the

[1] Institute of Organic Training and Advice

organic sector, recent meta analyses, the Organic Farm Management Handbook and the organic module of the Defra Farm Business Survey (de Ponti *et al.*, 2012; Seufert *et al.*, 2012; Lampkin *et al.*, 2012; Jackson *et al.*, 2008; Moakes and Lampkin, 2009, 2010, 2011; Moakes *et al.*, 2012, 2013, 2014).

In a second phase the effect of "high" and "low" nitrogen fixation thresholds was assessed to determine the influence on N availability and productivity within each rotation and soil/rain class. Crop offtake data (kg N ha^{-1} yr^{-1}) were also analysed for treatment effects (i.e. effect of soil type and rainfall) using one-way analysis of variance (ANOVA). Results from the yield adjustments and statistical analysis can be seen in Appendix B.

2.3 Development of production impact model

In the next phase of work, yield data derived from the NDICEA-based rotation modelling were applied within a large-scale optimisation model, in order to answer the research question relating to the amount of food that would be produced under a 100% organic agriculture (see section 1.2). For the completion of this task, the author of this study constructed an Optimal Land Use Model (OLUM) within the GAMS[2] programming language. A basic-working model was established over an intensive period of activity at the University of Reading from July to September 2014, and the model refined over the months and years that followed. During this period regular discussions on the modelling approach were held with Mr. Philip Jones, Senior Research Fellow at the Centre for Agricultural Strategy at Reading. During these discussions it was deemed appropriate to address the limitations of earlier studies with particular regard to calculations addressing feed availability, nitrogen supply and offtake and rotational constraints within organic systems (see section 3.7 Previous attempts to explore the impacts of up-scaling organic).

[2] General Algebraic Modelling System

The OLUM followed a linear programming approach, where an objective variable in maximised subject to constraints. The basic structure of the model is described in equation 2 below.

$$\text{Maximise: } Z = \sum_{ij=0}^{n} C_{ij} \times X_{ij} \tag{2}$$

$$\text{Subject to: } Ax_{(ij)} \leq b;\ x_{(ij)} \geq 0$$

The objective function, which was maximised, was aggregate food output (Z) as metabolisable energy. C is the energy output of organic agricultural products (i) on a range of soil/rain classes (j) and X is the activity scalar (crop areas or livestock numbers). A represents the input and resource requirements associated with diverse agricultural activities (x) and b is the resource endowment and input availability vector (e.g. manure-N, land by site class). For more detail on the mathematical structure of the model see Chapter 6 and Appendix C.

An overarching assumption within the OLUM was that the current agricultural area in England and Wales will remain the same post-conversion to organic management (i.e. there would be no expansion or contraction of the total farmland area). An additional assumption concerned the area of land by farm-type, which was presumed to remain constant following a switch to an organic scenario. This assumption follows evidence suggesting that dominant farming enterprise(s) tend to remain in-place post-conversion (Howlett *et al.*, 2002; Langer, 2002). The nine Defra Robust Farm Types were therefore used as a basis for the construction of nine representative organic farms, i.e.:

1. Dairy
2. Less Favoured Area (LFA) Grazing Livestock
3. Lowland Grazing Livestock
4. Specialist Cereals
5. General Cropping
6. Specialist Pigs
7. Specialist Poultry
8. Mixed arable and livestock
9. Horticulture

Organic farm data within the Farm Business Survey (FBS, years 2009 to 2011) provided most of information required for the farm-type construction (in particular crop areas and livestock numbers). For some farm-types data gathered in previous research were used (e.g. Edwards 2002, Leinonen *et al.* 2012a,b, for specialist pigs and poultry) as these systems are under-represented or missing from the organic sample within the FBS. Farm structure data were collected through email/telephone communication, through online sources (in particular the Defra website and via the online repository Organic Eprints) and through consultation with experts during a work/study period at the Soil Association head offices in March/April 2016.

Crop yield data were derived from the NDICEA-based assessments described above. Yield data for some crops (i.e. those excluded from the typical rotations described in Table 2.2) were extracted from published studies, e.g. oilseed rape yields were drawn from a survey of French organic farmers (Valantin-Morison and Meynard, 2008). Productivity estimates for permanent grassland and rough grazing were derived from a regression model described in Williams *et al.* (2006) which estimates grassland productivity for the site classes described in Brockman (1994) - see Appendix B. Livestock productivity data were extracted from the Organic Farm Management Handbook (Lampkin *et al.*, 2012) and industry-focussed research projects that include data on organic-farms (e.g. AHDB Dairy, 2014).

Current land area(s) for each farm type (in hectares) were then derived from the June Agricultural Census (2010) on a 5km^2 basis for each NUTS1 region of England (i.e. North East England, North West England, Yorkshire and the Humber, East Midlands, West Midlands, East of England, South East England and South West England) and for Wales. The farm type areas were combined with rainfall and soil type data derived from the UK Met Office and the National Soils Resources Institute (also reported on a 5km^2 basis over England and Wales). A land cover map was then used to remove extraneous areas from the dataset (e.g. urban conurbations, bracken, sea estuary, woodland). The result of this process was a breakdown of agricultural areas by rainfall band, soil type and farm type within each NUTS1 region, as illustrated in Figure 2.3:

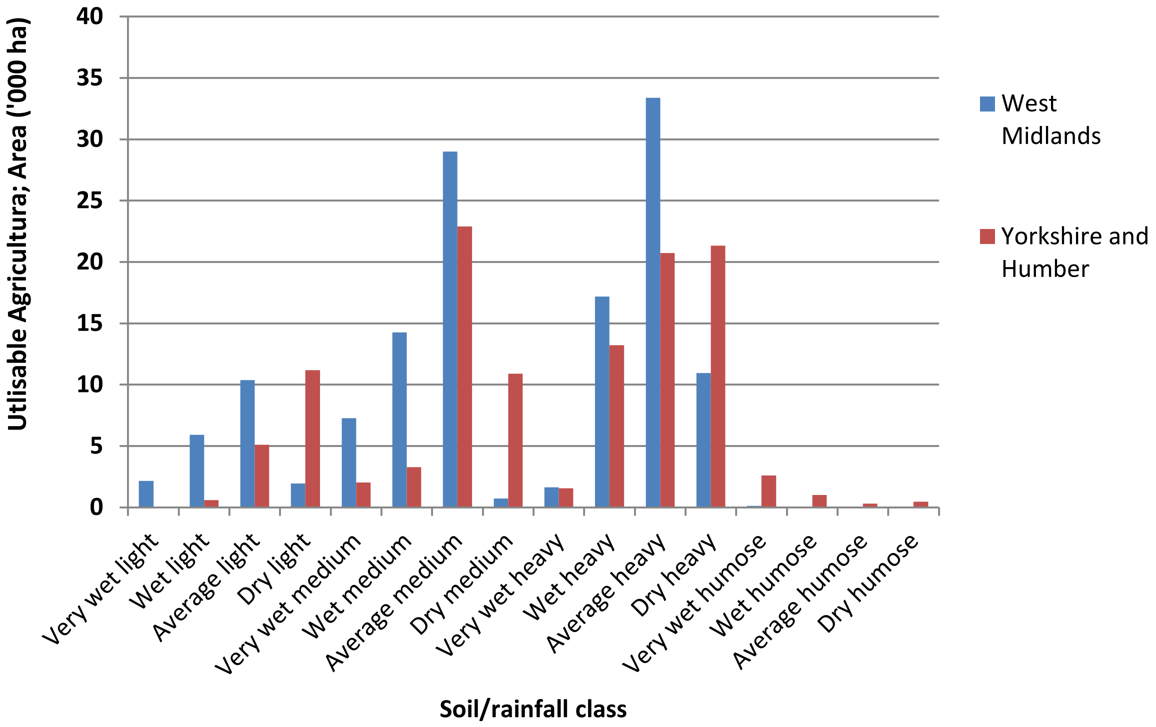

Figure 2.3. Land area estimates by land class for the NUTS1 regions 'West Midlands' and 'Yorkshire and Humber' for the 'Mixed' Robust Farm Type. A combination of data sources allowed for the estimation of the land areas for each farm type applied in the OLUM.

Limits on the availability of land within each soil/rainfall class (and the associated crop yield, derived from the NDICEA based yield adjustments) were set through the application of land-use constraints within the OLUM. The model was then tasked with populating each NUTS region with the same area of each farm type as the 2010 situation, maximising the food energy produced, as described in the objective function (equation 2 above).

Following a 'base-run' (i.e. a 'best-guess estimate of what a 100% organic England and Wales would produce) a range of scenarios were assessed by adjusting key model parameters. The scenarios were based on a review of the marginal value for each constraint applied in the OLUM[3] through an assessment of the GAMS output files. By reviewing the marginal values it was possible to identify constraints that had the greatest influence on the objective function (i.e. maximising food production). Separate scenarios were also implemented for low and high rates of biological nitrogen fixation, as these are highly uncertain (Döring *et al.*, 2013; Galloway, 1998) and higher/lower estimates significantly affected crop yields within the NDICEA based adjustments (see yield adjustment results in Appendix B). Following the implementation of a range of scenarios, a 'combined' scenario that allowed for a reduction of fallow areas and the recycling of imported food residue was selected for the next stage of work (i.e. the environmental impact assessment) as this scenario markedly improved the levels of food production under organic management and was therefore deemed to be the most realistic (total food production was still considerably reduced compared to a conventional baseline).

2.4 Environmental impact modelling

In the final modelling phase, a Life Cycle Assessment (LCA) model was applied to determine the environmental impacts of a 100% organic scenario. The Cranfield Agri-LCA models provided the basis for this work (Williams *et al.* 2006) and an introduction to their function and application was provided in June 2016.

Following this introduction, default crop areas by soil and rainfall class were adjusted within the Agri-LCA interface using OLUM-derived data for each of the crops assessed in this study (e.g. areas of winter wheat on each soil and rainfall class). The adjusted cropland areas fed into the fossil fuel and N-loss modelling within the Agri-LCA and the associated CO_2 and N_2O calculations. A range of industry data sources were also applied within the LCA models to better-represent organic production systems and to estimate the impacts of a non-organic baseline (i.e. based on 2010 levels of production, see Appendix E for a

[3]The marginal value can be considered to be the amount by which the objective function would change if the right hand side would be moved one unit (McCarl *et al.* 2014)

list of data sources applied). These industry data were derived through email exchanges, telephone calls and/or through online resources. The information sources collected fed-into parameters on cropping and livestock production (e.g. cultivation requirements, milk yields, replacement rates, cattle liveweights) within the Agri-LCA and were used to adjust the impact calculations (e.g. fossil fuel use, feed-intake and N excretion from livestock). The result of this process was an updated estimate of the greenhouse gas emission and fossil-energy use impacts associated with a range of organic crop and livestock products. Total impacts for each product type were expressed per kilogram of product as illustrated in Chapter 7. The impact of the 100% organic scenario was then assessed by combining the per kilogram estimates with the total organic food production estimates, derived from the OLUM. The impacts of a non-organic baseline were also assessed using default conventional production data from the Agri-LCA and national production records (e.g. Defra cereal production datasets).

Overseas production impacts were then assessed through an extraction of data from peer-reviewed literature (mainly other LCA-based studies), data from research project reports and official Government and industry statistics (see Chapter 7 and Appendix E). This allowed for an estimate of the total overseas land-area currently utilised in the production of imports, and for a calculation of the additional land required under an organic scenario (more overseas land was required under the organic scenario due to a reduction in domestic productivity). The greenhouse gas impact associated with the cultivation of additional land was determined through an application of country-specific land use change values within LCA guidelines (British Standards Institute, 2011). Carbon sequestration estimates for the organic scenario were also included in the environmental impact assessment, with rates of sequestration derived from a recent meta-analysis, and applied to the eligible organic area (Gattinger *et al.*, 2012). A range of sequestration values reported within the same meta-analysis were applied within a sensitivity analysis as described in Chapter 7. Figure 2.4 provides an overview of the steps taken within the environmental impact assessment of the 100% organic scenario.

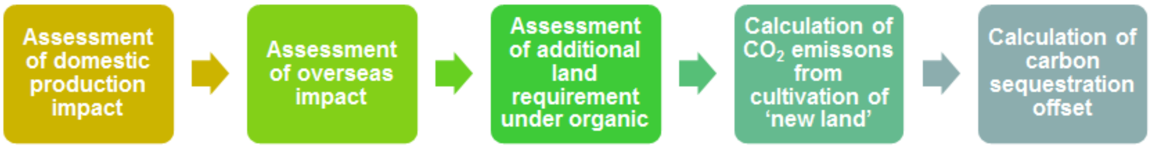

Figure 2.4. Flow chart depicting stages of work implemented in the 'scaling-up' of environmental impacts for the 100% organic production scenario

Energy efficiency ratios were also constructed for each of the crops and livestock products considered in Chapter 7 by converting the energy content of foodstuffs to metabolisable energy (ME) using standard composition tables (McCance, 2002) and data within a separate study comparing the efficiency of livestock production systems (Wilkinson, 2011). Total energy outputs were then divided by the total fossil-fuel ME-use for each commodity, using data derived from the Agri-LCA, in order to calculate a production efficiency ratio by product type (see Table 7.6).

2.5 Timeline of activity

An overview of the stages of work and their approximate time of application is provided in Table 2.4 below:

Table 2.4: overview of the tasks applied over duration of part-time PhD thesis

Task Description	2012				2013				2014			
	Q1	Q2	Q3	Q4	Q1	Q2	Q3	Q4	Q1	Q2	Q3	Q4
Literature reviews 1			*x*	*x*	*x*	*x*	*x*	*x*				
NDICEA validation and long-term trial modelling								*x*	*x*	*x*	*x*	*x*
OLUM construction and modelling of 100% organic scenario											*x*	*x*

Task Description	2015				2016				2017			
	Q1	Q2	Q3	Q4	Q1	Q2	Q3	Q4	Q1	Q2	Q3	Q4
NDICEA-based yield modelling	*x*	*x*	*x*	*x*	*x*							
OLUM construction and modelling of 100% organic scenario	*x*	*x*	*x*	*x*	*x*	*x*	*x*					
LCA of results from modelling						*x*	*x*	*x*	*x*			
Literature reviews 2							*x*	*x*	*x*	*x*		
Thesis writeup and submission							*x*	*x*	*x*	*x*		

Literature reviews in the early stages of work focussed on the identification of previous studies and suitable modelling framework(s) and the completion of a review exploring the energy efficiency of organic farming systems (Smith *et al.*, 2015). The most intensive period of work took place between June 2014 and December 2016. During this phase the OLUM model was constructed from scratch and continuously refined. At the same time the NDICEA model was applied to produce a set of organic crop yields adjusted by soil class and rainfall zone. The bulk of environmental LCA-work took place in mid-to-late 2016 whilst reviews exploring the broader impacts of organic cropping and livestock system production commenced in the same period and were finalised in 2017. The following chapters contain more information on the methods used and results from their application.

CHAPTER 3. GENERAL LITERATURE REVIEW

This chapter provides some general background information on the organic sector and an overview of previous work addressing the same or similar research questions as those described in Chapter 1. A range of models are also considered, along with examples of their application in published material. Each model is assessed against a range of success criteria, defined within the context of this study. The model comparison forms the basis for the selection of an assessment framework.

3.1 Organic farming – an introduction

Organic agriculture is an approach to food production that aims to create environmentally, socially and economically sustainable farming systems that rely on farm-sourced or local resources and ecological processes (Lampkin *et al.*, 2012; Shepherd *et al.*, 2003; Stockdale *et al.*, 2001). As a result, a central tenet of the organic approach is an avoidance of synthetic fertilisers or pesticides. Instead organic farms rely on organic fertiliser and crop rotation to promote soil fertility (Committee on Climate Change, 2010) in particular through the use of clovers and other legumes, which provide the main N-input to organic systems via biological nitrogen fixation.

Increased diversity of cropping and livestock can also be observed on organic farms (Stockdale *et al.*, 2001) with intercropping and under-sowing often being used to maintain a level of soil cover throughout the year. Livestock also play an important role within organic production, providing a source of mobile fertility through manures and slurries whilst making use of the fertility building ley within organic rotations (Lampkin, 2002). Such approaches help to develop a 'closed-cycle' with regard to nutrient use (Novak and Fiorelli, 2009; Stolze *et al.*, 2000) although it is recognised that this cannot be absolutely attained (Lampkin et al., 2015). The term organic is therefore not directly related to the type of inputs used but instead refers to the concept of the farm as an organism, through which all of the components of the farm interact and influence each other within a coherent whole (Steiner, 1924). This positivist approach distinguishes organic farming from other modes of production that focus on single aspects, such as individual crop diseases and how they can be treated (Baars, 2002).

3.1.1 Development of the organic sector

Scientists such as Sir Albert Howard and Lady Eve Balfour were early proponents of organic farming, and their work with alternative husbandry and cropping systems placed a particular emphasis on the links between the way that food is produced, food quality and human health. The Haughley Experiment, established by Lady Eve Balfour in 1939, was one of the first comparisons of organic and conventional farming, and the results highlighted the importance of healthy, fertile soils in the production of healthy crops and livestock, and the subsequent link with human health (Balfour, 1943). The early organic movement therefore focussed strongly on issues of human health, diet and nutrition and the promotion of soil fertility through the use of composts and other organic fertilisers. The issue of pesticide use did not come to the fore until the publication of Rachel Carson's Silent Spring in the early 1960s, which generated widespread public concern (Carson, 1963; Stockdale *et al.*, 2001). In addition, in the late 1960s broad societal changes led by student movements resulted in an increased emphasis on the social and cultural aspects in organic agriculture, although there was a clear tension between the conservative founders of the organic movement and the more radical younger generation. During the 1980s and 1990s issues of biodiversity conservation, animal welfare and social justice relating to fair trade with developing countries gained importance and the potential of organic agriculture to contribute to rural development was highlighted (Stockdale *et al.*, 2001).

It has therefore taken some time for the ideas behind organic agriculture to develop into a consistent and unified concept that is adhered to worldwide. However, strong unifying principles now link the wide range of farming systems within the organic sector (Stockdale *et al.*, 2001). This process has been assisted by IFOAM (the International Federation of Organic Agriculture Movements) who have defined organic agriculture along the lines of four key principles:

- **Health:** Organic agriculture should sustain and enhance the health of soil, plant, animal, human and planet as one and indivisible
- **Ecology:** Organic agriculture should be based on living ecological systems and cycles, work with them, emulate them and help sustain them
- **Fairness:** Organic agriculture should build on relationships that ensure fairness with regard to the common environment and life opportunities
- **Care:** Organic agriculture should be managed in a precautionary and responsible manner to protect the health and well-being of current and future generations and the environment

Source: IFOAM website: http://www.ifoam.org/about_ifoam/principles/index.html

The principles defined by IFOAM are implemented locally by national or regional certification and inspection organisations, although a set of basic standards provide criteria that accredited certifiers in each country must fulfil (IFOAM, 2002).

3.1.2 Organic farming policies, regulations and standards

Individual countries have set out policy initiatives defining and supporting the development of the sector since the mid-1980s, and Padel and Lampkin (2007) highlight that this support was provided in three key contexts. Firstly there was a relatively short-lived idea that low yields from organic farming might help reduce overproduction, encouraged by ‘Pillar 1’ support payments within the Common Agricultural Policy (CAP) following food shortages after the Second World War. Secondly the organic sector was an infant industry, support for which could be justified in terms of expanding consumers choices and lastly it was seen as providing ‘public goods’, i.e. environmental, social and other benefits to society that are not covered by the price of the food sold.

Individual certification bodies have also developed minimum standards for organic production, both to protect consumers from fraud and to protect producers from unfair competition. The first organic standards were set through ‘basic-norms’ defined by the biodynamic movement[4] in Germany in 1928. These norms set requirements for producers who wished to use the ‘Demeter’ name on their products, and focussed on the use of seed and the length of conversion periods (i.e. the time period since the land was treated with manufactured fertilisers). Since the 1920s and 30s a plethora of national and international organic standards have been developed by certification companies. In early years these often consisted of basic guidelines, with standards becoming more prescriptive and rule-based over time. IFOAM have played an active role in helping standards to become more consistent, in particular through the publication of ‘basic-rules’ (i.e. minimum requirements) for organic standards that have helped to facilitate international trade (Schmid, 2007).

More recently, the development of private organic standards and national policies led to the formation of Regulation (EEC) 2092/1991, which was introduced in 1992 to protect organic farming by ensuring fair competition and transparency at all stages of production and processing (Padel and Lampkin, 2007). Both the European Regulation and the US Department of Agriculture’s National Organic Programme (NOP) have strongly influenced the development of the sector worldwide, in particular by improving the credibility of organic products in the marketplace (Padel and Lampkin, 2007).

In line with the European Action Plan for organic food and farming, the European Commission began the process of a total revision of its organic regulation in 2005. The majority of the rules related to production remained

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[4] Biodynamic farming has many similarities with organic agriculture although its origins can be traced back to a single set of lectures given by the Austrian philosopher Rudolf Steiner (1924). On biodynamic farms a stronger emphasis is placed on the importance of the farm as an organism, the role of ruminant livestock, and the use of preparations derived from manure, herbs and minerals, for improving soil and crop health. For more information see: Wistinghausen, E., Sattler, F., 1992. Biodynamic Farming Practice. The Biodynamic Agricultural Association, BDAA, Stroud, England.

unchanged in the new regulation, but the number of derogations[5] was reduced and a framework for strictly regulated regional flexibility introduced. The new standards for organic production in the European Union were published in 2009[6].

The updated European organic standards also provided an organic logo, which helps to convey the organic character of the product (Figure 3.1). Use of the logo is now compulsory on all pre-packed organic products produced and/or sold in the EU. Recent research suggests that although the logo has improved clarity to consumers, in the UK there is still some way to go before it is accepted and understood widely (Gerrard *et al.*, 2013).

Figure 3.1: Organic logo of the EU. Source: Article 57 of standard 834/2007

The regulations introduced by the European Commission ensure that organic certification is communicated to consumers through the protected term ‘organic’ (or an equivalent protected terms in other languages), and that products can only use the term ‘organic’ (or one of the other terms protected by the Regulation) on the label if they have undergone inspection/certification. Organic products also have to show the name and number of the approved control body.

The EU organic regulation was subjected to a second review which began in 2012, and the European Commission published a proposal for a new, updated organic regulation in 2014. The proposals have been poorly received by IFOAM and its members due to inadequate stakeholder consultation, technical shortcomings and inconsistencies concerning import rules and the improvement

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[5] Derogations refer to special permissions which may be requested by a producer or processor who wishes to use inputs that are normally prohibited in organic agriculture (e.g. manufactured pesticides, conventionally grown seeds or certain veterinary medicines). Requests are normally made in writing and addressed to the certification body operating in an individual country.

[6] The latest standards are entitled ‘’Council Regulation (EC) No 834/2007 of 28 June 2007 on organic production and labelling of organic products and repealing Regulation (EEC) No 2092/91’’. The Commission regulation EC No 889/2008 of the 5th of September 2008 lays down in more detail the rules for the implementation of regulation 834/2007. Both regulations are available on the website of the European Union (see: http://eur-lex.europa.eu/en/index.htm)

of control systems (IFOAM EU Group, 2015; Padel and Woodward, 2014). Whilst the new organic regulation is expected to enter into application in 2018 or 2019, an extensive consultation process is currently ongoing to resolve such issues.

Despite such setbacks, the European organic movement has evolved from being opposed to agricultural policy developments, to being fully subsumed as part of their development. This has led some to state that the organic movement has lost control of its own destiny and that policy support has "pushed the sector towards protecting the environment, rather than producing food in a more sustainable manner" (Tovey, 1997). Lampkin and Padel (2007) highlight that in some cases "organic farming has indeed just become one of many agri-environmental schemes" although they go on to highlight that "the introduction of single European or national regulations, as opposed to the plethora of standards, has helped to reduce confusion and inspire confidence in consumers".

3.2 Growth and current status of the organic sector

According to a recent IFOAM and FiBL (Swiss Research Institute of Organic Agriculture) survey there are 43.7 million hectares of organic agricultural land worldwide, comprising 1.0% of all agricultural land. The highest shares of organic land are found in Oceania (4.1%) Europe (2.4%) and within the European Union (5.7%). However many countries within Europe have much higher percentages, for example Austria (19.4%), Liechtenstein (30.9%) and Sweden (16.4%) (Willer and Lernoud, 2016).

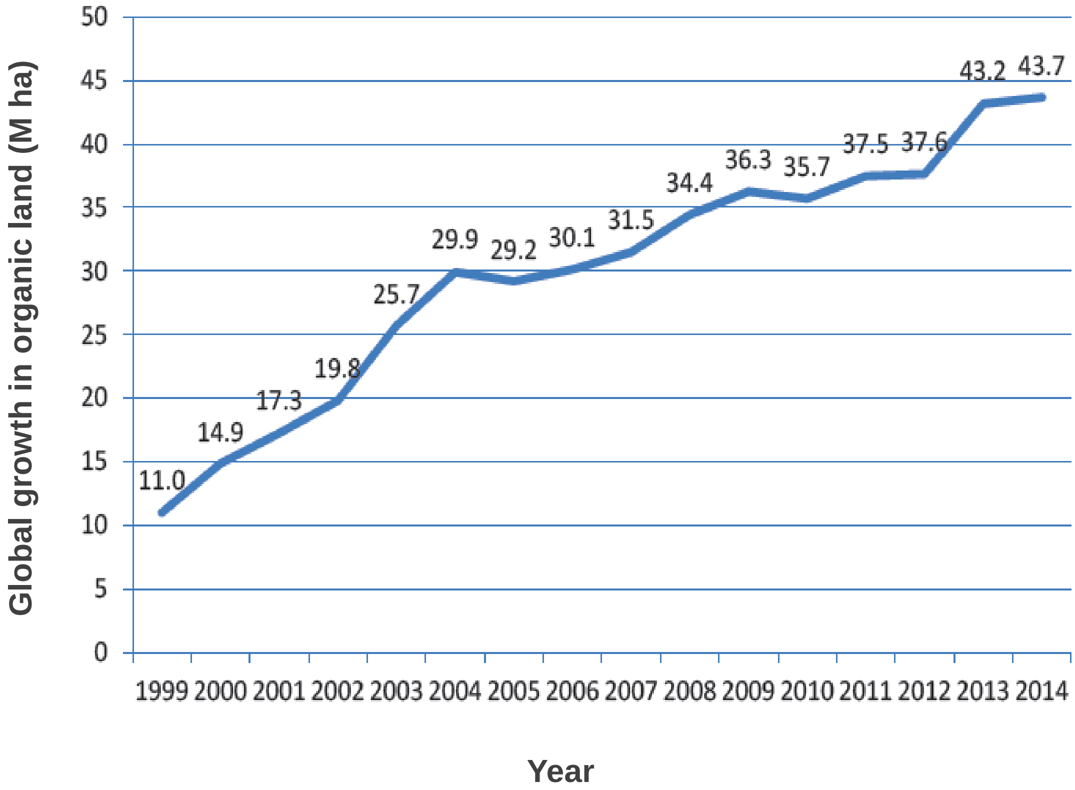

Figure 3.2: Growth in global organic agricultural land 1999-2014. Source: Willer and Lernoud (2016)

The dramatic growth in the organic sector over recent years (Figure 3.2) can be partially attributed to growing policy support, in addition to an increasing consumer demand, which largely stems from a desire to buy "healthy" organic food (healthiness is often associated with the absence of chemical residues, as well as a higher nutritional value, amongst organic consumers, Padel and Lampkin, 2007; Seufert *et al.*, 2017). Animal health, food safety and economic problems in the non-organic sector have also led to increases in the consumption of organic food products (Padel and Lampkin, 2007). In addition some countries, such as Sweden, have set specific targets for a large increase in organically farmed land areas, to help mitigate the environmental impact of non-organic farming systems (McNeill *et al.*, 2005).

Global organic sales have also been increasing steadily over the last 25-30 years, reaching a combined value of 80 billion US dollars in 2014. The size of the global market has expanded over five-fold since 1999, making it one of the fastest growing food sectors (Sahota, 2016; Seufert *et al.*, 2017). Markets in North America and Europe comprise 91% of global revenues (Sahota, 2016) highlighting a current disparity with the location of the majority of organic producers (over two-thirds of organic farmland is in developing countries where the focus is on exports, Sahota, 2016).

3.3 Organic agriculture in the UK

The amount of fully converted land in the UK is about 508,000 hectares, representing 2.9% of the total farmed area, although the proportion of land that is farmed organically varies widely by region, from highs of 10.7% in south-west England and 8% in Wales to 1.5% or less in Northern Ireland, East Midlands, East England, Yorkshire and Humberside (Soil Association, 2012, 2014). The dominant type of organic land is permanent and temporary pasture, covering 84% of organic land, with cereals being the next largest area, covering 7.6% of the total (Defra, 2017).

Over half of all organically managed land within the United Kingdom is in England, with the South West region having the highest proportion of both crop and livestock producers (Defra, 2017). There was a 5% drop in the amount of organic land in the UK in 2016 (Soil Association, 2017), as some farmers reacted to market conditions and reverted to non-organic production, particularly as a result of rising imported feed costs and uncertainty over organic support payments following the British referendum on EU membership (Lampkin *et al.*, 2017).

Within the UK the main certification bodies are the Soil Association, Organic Farmers and Growers and the Biodynamic Association. In many cases, standards set by these control bodies go above and beyond the minimum standard prescribed by the European Commission. For example the Soil Association standards require smaller flock sizes for chickens on animal welfare grounds. The Soil Association is currently the largest certifier of organic products in the UK.

Support payments for organic conversion in the UK are currently available under the Rural Development Programme 2014-2020 and through Countryside Stewardship options. Following the EU referendum result, it is currently unknown whether new applicants for agreements starting in 2018 will be eligible for payments. Current support payments for organic farming in the UK are amongst the lowest in Europe (Figure 3.3) and a lack of public investment in this area is a contributory factor to the current small size of the sector (Lampkin *et al.*, 2017; Sanders *et al.*, 2011) .

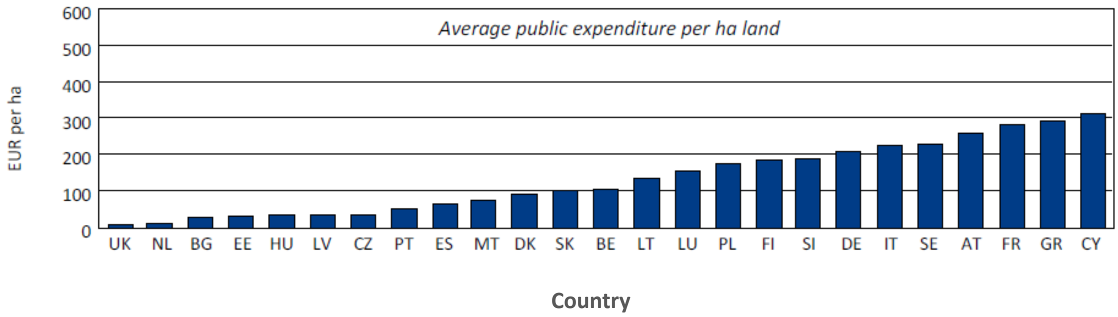

Figure 3.3: public expenditure for organic farming support across EU-25. Source: Sanders *et al.* (2011)

3.3.1 The UK organic market: recent developments

Although the beginning of the recession in 2008/9 led to a considerable decline in organic sales, the current UK organic food market represents a position of considerable recovery (Soil Association, 2010a, Mintel, 2013). In 2016, total sales of organic products in the UK increased for the fifth consecutive year, with total sales increasing by 7.1%, partly as a result of the devaluation in Sterling following the UK referendum on EU membership, which has led to increased organic food exports (Soil Association, 2017). Sales of organic milk increased by 3.2% in volume terms, after a decrease of 2.2% in 2015, although more recently sales have been negatively affected by the low price of non-organic milk (Soil Association, 2017).

Red meat sales also increased in 2016, which may in-part may relate to new evidence from Newcastle University on the nutritional benefits of organic meat in terms of fatty acid composition and concentrations of beneficial minerals and antioxidants (Soil Association, 2017; Średnicka-Tober *et al.*, 2016). Pig and poultry producers have been struggling with higher feed and energy prices, although poultry numbers rose by 6.7% in 2015 and sales of organic eggs and poultry meat have increased by 3.1% and 4.1% respectively, partly as a result of increasing consumer concerns over animal welfare in non-organic production (Soil Association, 2016). The Soil Association also highlight that the representation of organic products in supermarkets is increasing, particularly as the 'discount' supermarkets are now stocking more organic products to attract consumers (Soil Association, 2017) although dynamic growth in the home-delivery and independent retail sectors is also driving the current expansion. The future prospects for the sector are encouraging with a 5% expansion in the UK organic sector's market value predicted for 2017 and with the UK-based Organic Trade Board being awarded over 10 million Euros of European funding to promote retailer uptake of organic products (The Organic Trade Board, 2017, Soil Association, 2017). Rising (non-organic) food prices are also expected to contribute to the further development of the sector in the UK as conventional producers are encouraged to switch to more extensive methods of production to improve net-margins (Soil Association, 2017).

Despite increases in sales, the UK's organically farmed land area has been decreasing in recent years (Figure 3.4), primarily as a result of rising feed costs and uncertainty over the future profitability of the sector (Daneshkhu, 2016, Rustin, 2015.).

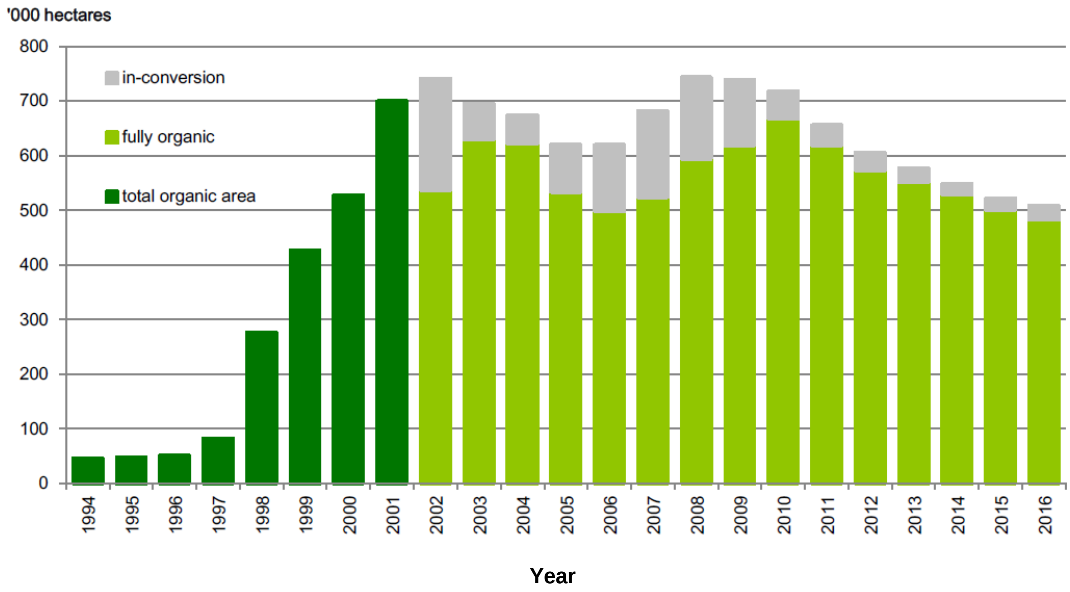

Figure 3.4: area of UK land in-conversion and fully organic (Defra, 2017)

Considerable purchasing barriers also still exist in relation to organic products, with cost the biggest factor although many also consider the term ‘organic’ and it’s underlying principles to be vague. A third of organic consumers would therefore like to see more information on what ‘organic’ really means for each product (Mintel, 2013). Fresh fruit, vegetables, dairy products and baby food are the most popular organic foodstuffs in the UK, with the absence of pesticide residues a particularly important factor in determining the popularity of organic fresh products and the success of one or two key brands driving the expansion of dairy products and baby food (Mintel, 2013).

3.4 Characteristics of organic farming systems

As mentioned in section 3.1, the organic approach focuses on the whole farm system, rather than individual components. In this sense a conversion to organic agriculture can involve a significant restructuring of the farm, rather than an adaptation of current practice (Stockdale *et al.*, 2001) although the dominant enterprise usually remains in-place post conversion (Howlett *et al.*, 2002; Langer, 2002). Mixed farms are also more commonly found within the organic sector, although specialised (e.g. stockless arable) farms do also occur (El-Hage Scialabba and Hattam, 2002; Smith *et al.*, 2011).

Stockdale *et al*. (2001) state that common characteristics of organic farms include the following:

- A focus on practices that can promote soil fertility and protect or enhance soil organic matter (SOM)
- The adoption of practices that can conserve natural habits and wildlife within agricultural areas
- The use of ‘natural’ methods of pest, disease and weed control, including crop rotations, the use of biological control from natural predators and resistant varieties
- Extensive livestock management focusing on meeting the behavioural and physiological needs of animals with particular regard to housing and nutrition/diet

In addition a study by Gabriel *et al*. (2009) found that the highest concentrations of organic farmland in the UK are in areas less suited to arable farming, characterized by high altitude and slope, and that typically organic farms are small and likely to be ‘mixed’ or ‘dairy’ farm types. Padel and Lampkin (2007) also found that organic farms subscribing for support schemes in Europe are generally low intensity livestock or mixed production systems located in marginal areas.

In the following sections, the key characteristics of crop and livestock enterprises within the organic sector are explored in more detail.

3.4.1 Crop production

Crop production within organic farming is characterised by a diversity of cropping patterns, compared to non-organic farming systems, with the objective of minimising weed, pest and disease problems and the closing of nutrient and organic matter cycles (Lampkin *et al.*, 2012). Although the ideal of self-sufficiency within organic systems can never be fully achieved as exports are rarely balanced with the return of human waste to the farming system (in particular as the use of human sewage on organic farms is currently prohibited under organic standards, Soil Association, 2010b) it is seen as something to be aimed for, and any external inputs should ideally be derived from renewable sources (although this can be hard to achieve in practice). In general, only fertilisers that release nutrients slowly over time through soil microbial processes or weathering are allowed to enter the farm. Where a need for supplementary nutrients is proven and cannot be addressed within the farming system or by re-design of the rotation, application of supplementary fertilisers can be made, with rock phosphate and a range of potassium minerals being two common sources (Stockdale *et al.*, 2001).

Lampkin *et al.* (2012) highlight that there is no standard organic crop rotation, as conditions and requirements vary greatly from site to site and with market fluctuations, however Table 3.1 provides some guidance on the typical proportions of crop categories on organic farms:

Table 3.1: Proportions of crop categories within organic rotations. Source: Lampkin *et al.* (2012)

Crop category	Min (%)	Max (%)
Forage and green manure legumes and leys	20	100
Cereals	0	50
Other fodder crops	0	33
Grain legumes & oil seeds	0	25
Roots & vegetables	0	75

As can be seen from Table 3.1, medium to long-term grass-clover leys are included in organic rotations to a greater or lesser extent, in order to build fertility and for grazing and fodder production. For stockless systems, short/medium term green manures (e.g. *Trifolium, Medicago and Vicia* spp.) are used instead, and these may be cut and mulched or incorporated directly into the soil (Stockdale *et al.*, 2001). Where grass-clover leys are cut and mulched, however, the mulched crop acts as fertiliser and can promote grass-growth at the expense of clover (Dahlin and Stenberg, 2010) resulting in lower clover contents in older leys (Nykänen *et al.*, 2000). Lampkin *et al.* (2012) also highlight that the short-term green manures/leys common within organic stockless organic systems may not provide enough N through biological fixation to support the following crops (i.e. at the 20% level in Table 3.1) and that the ‘Max’ values for each of the cereals, grain legumes, other fodder crops and vegetables may create problems for specific diseases, build-up of weeds and reductions in organic matter.

Partly as a result of the restriction on synthetic input use, individual crop species and varieties within organic systems are chosen for their adaptability to local soil and climate conditions and resistance to local pests and diseases (El-Hage Scialabba and Müller-Lindenlauf, 2010). Organic farmers also develop more resilient systems through growing a mix of crop varieties, avoiding the vulnerability to extreme weather events created by monocultures (Smith and Lenhart, 1996) and making more efficient use of available nutrients (Zhang and Li, 2003).

3.4.2 Livestock production

In line with the aim of creating a closed-system, plant and livestock production are integrated on organic farms to optimise nutrient use and recycling, with grass/clover from leys and arable crops providing feed for livestock and the livestock providing a mobile source of fertility for the grassland and crops (Watson *et al.*, 2002). Ruminants therefore play a vital role in many organic systems, by making productive use of fodder legumes, which also provide the main source of nitrogen to the farm. This approach aligns with organic standards which dictate that at least 60% of the diet (on a dry matter basis) for ruminants shall be forage based (European Commission, 2010). Organic farming standards also require that feed brought into the holding for livestock is grown organically, although organic pigs and poultry units are allowed to import a small-amount of non-organic protein feed (European Commission, 2010; Hermansen, 2015). The use of pure nutrients and additives is also limited to “natural” ingredients and levels of feeding are determined by an animal’s minimum needs rather than maximum production levels (Stockdale *et al.*, 2001). This stipulation can lead to negative effects in terms of meeting animals’ requirements, particularly in the poultry sector where the prohibition on the feeding of synthetic amino acids can contribute to worse feed conversion, N excretion, and mortality rates compared to conventional free-range or fully-housed systems (Dekker *et al.*, 2012; Leinonen and Kyriazakis, 2016; Steenfeldt and Hammershøj, 2015).

Many organic farms will also aim to retain as closed a herd/flock as possible, and thereby reduce the risk of buying in disease whilst encouraging selection on the basis of an individual farm's needs (Stockdale *et al.*, 2001). This approach may lead to missed opportunites in terms of the 'hybrid-vigour' that can be obtained from introducing new genetics through cross-breeding (Buckley *et al.*, 2014). A lack of suitable breeds can also pose an issue in the organic pig and poultry sectors which tend to rely on modern breeds that are not always suited to free-range/outdoor conditions (Nauta, 2001). Organic farms also aim to apply species-specific husbandry with consideration for animals' ability to express their natural behaviour. Organic farming therefore aims to go beyond the alleviation of pain, fear and hunger as set out within the RSPCA 'Five Freedoms' (Main *et al.*, 2001) with the aim of respecting the integrity of the animal. The focus is therefore on efficiency, rather than maximisation of output, with longevity being an important factor within organic breeding/selection (Van Diepen *et al.*, 2007). Evidence also shows that the use of dual-purpose breeds on organic dairy farms is greater than within the non-organic sector (Van Diepen *et al.*, 2007). Feeding and management regimes must also be geared towards the species and class of stock being fed, taking account of nutritional adaptations and requirements (early weaning is discouraged under organic management, Stockdale *et al.*, 2001).

With regard to animal health, organic farms take a preventative approach, through appropriate housing, and management techniques (e.g. rotational grazing, Soil Association, 2008). Whilst the application of these approaches can help to reduce the disease challenge and increase the resistance of host livestock, organic standards emphasise the need to treat any sick animal with the best available means (Stockdale *et al.*, 2001). Despite the adoption of such preventative measures, internal and external parasites can pose particular problems on organic farms, and inherent shortfalls in trace-elements in animal diets can occur through a greater reliance on home-produced forages (Stockdale *et al.*, 2001). Mastitis is recognised as the dominant health problem for organic dairy herds (Haskell *et al.*, 2009) and higher mortality rates have been found in organic poultry systems as a result of increased metabolic energy requirements, predation pressures and a greater incidence of feather pecking as a result of untrimmed beaks (Dekker *et al.*, 2012; Leinonen *et al.*, 2012a, b). The practices of intentional mutilation of livestock in the form of castration, dehorning, and tail and teeth clipping vary according to livestock type, for instance virtually all organic dairy herds within the United Kingdom are de-horned (Roderick *et al*. 1996 in Stockdale *et al*. 2001) and tail docking of lambs is commonplace, whereas teeth clipping and tail-docking of pigs is prohibited under certain organic standards (Soil Association, 2008).

3.5 Organic agriculture's impact on climate change

An overview of recent work exploring the impact of organic production on greenhouse gas emissions and soil carbon sequestration is presented in the following section. A separate, detailed assessment of fossil energy efficiency within a range of organic systems is presented in Chapter 4.

3.5.1 Greenhouse gases from agriculture – an overview

The Fifth Assessment Report of the IPCC states that greenhouse gas (GHG) emissions from agriculture, between 2000 and 2010 were estimated to be 5.2 – 5.8 $GtCO_2$ eq yr^{-1} (i.e. 10-12% of global anthropogenic emissions, Smith *et al.*, 2014). Moreover GHG emissions from deforestation, due to land conversion to crop or livestock production, account for c.12% of global GHG emissions (El-Hage Scialabba and Hattam, 2002; El-Hage Scialabba and Müller-Lindenlauf, 2010; Idel, 2013). When these elements are combined with food handling and processing activities, it is estimated that approximately one-third of global anthropogenic GHG emissions are due to agriculture (El-Hage Scialabba and Müller-Lindenlauf, 2010; Olesen, 2009). There is considerable uncertainty in the estimate, as a direct result of a lack of clarity over how emissions from clearing forests and cultivation of new land can be attributed to agriculture (Olesen, 2009). Despite this, agricultural systems clearly have a major role to play in reducing greenhouse gases. Moreover, agriculture contributes a disproportionate amount of high impact GHGs: approximately 47% and 58% of total anthropogenic emissions of methane (CH_4) and nitrous oxide (N_2O) respectively. Together, these two gases form over 90% of agriculture's contribution to global warming (not including Land Use Change, Foresight, 2011). For this reason, the following sections focus on the impact of organic practices on N_2O and CH_4 emissions.

3.5.2 Organic farming and nitrous oxide (N_2O) emissions

The IPCC estimates that 70% of the total GHG emissions from agriculture are associated directly with nitrogen (N) fertiliser, through a combination of the CO_2 and N_2O emissions arising from its manufacture and use (Powlson *et al.*, 2011). Agricultural N_2O emissions are forecast to increase by 35-60% as we approach 2030, due to increased fertiliser use and animal manure production (FAO, 2003). Organic systems avoid the emissions associated with manufactured N fertiliser, as the main source of N is biological nitrogen fixation, within the fertility building ley period of the crop rotation (Lampkin, 2007). Despite this, a global meta-analysis by Skinner *et al.* (2014) highlights that yield-scaled N_2O emissions under organic management are higher as a result of lower rates of production, although N_2O emission per hectare are generally lower as a result of lower N-inputs and livestock densities.

It has also been highlighted that the N use-efficiency of organic fertilisers is low, compared to equivalent quantities of inorganic fertiliser, as the N can often be released too late in the season, or even after the growing season has ended (McNeill *et al.*, 2005). Such effects are expounded by factors that are beyond the direct control of the farmer, such as the soil moisture, aeration and temperature, which affect the rate of N mineralisation from organic material (Novak and Fiorelli, 2009). These effects have led some authors to conclude that the N_2O emissions associated with the application of manure can be higher than for mineral fertiliser, depending on soil type (Petersen *et al.*, 2006; Rochette *et al.*, 2008; van Groenigen *et al.*, 2004). The amount of nitrogen supplied by legumes can also vary greatly and is difficult to predict, with ranges of <100 to 700 kg N ha^{-1} yr^{-1}, depending on the legume that is used (Ledgard, 2001). It is claimed by some that the total N budget on organic farms will therefore be inherently more unreliable than for conventional farms where the total N input (as manufactured fertiliser) is more easily controlled and predicted (Leach *et al.*, 2002).

In light of the unpredictable nature of the N sourced from legumes, improvements in N use efficiency on organic farms are particularly related to improvements in the use of manures as an effective nutrient resource (Novak and Fiorelli, 2009). This management must take into account the fact that nutrients from manure are available at a slower rate compared to mineral fertiliser, that fresh manure contains more readily available nitrogen than composted material, and that much of the available N in manure may be lost through the release of ammonia (NH_3) during composting (Larney *et al.*, 2006; Watson *et al.*, 2005). Manure analysis and improved timing of application may help to improve management by allowing for an enhanced prediction and uptake of available N (Novak and Fiorelli, 2009). Nitrogen leaching from manures can also be reduced, through the use of cover and catch crops and green manures (Niggli *et al.*, 2009; Stopes *et al.*, 2002) which can also improve soil structure, and enable significantly lower mobile nitrogen concentrations, further reducing N_2O emissions. Improvements in N utilisation within organic systems can also be made by breeding crops to improve their N-use efficiency (Wolfe *et al.*, 2008).

3.5.3 Organic farming and methane (CH_4) emissions

There have been few direct comparisons of methane generation between organic and conventional production (Lampkin, 2007) although evidence has shown that diets high in roughage will lead to higher rates of methane emissions than diets high in starch (i.e. diets with a high cereal content, Eckard *et al.*, 2010; Johnson and Johnson, 1995). The specifications within the European Commission's Organic Regulations for at least 60% of the dry matter in daily rations of herbivores to consist of roughage, fresh or dried fodder, or silage has therefore led some authors to conclude that a conversion to organic agriculture will result in higher levels of methane being emitted (de Boer, 2003). A reliance on high cereal diets can result in severe difficulties relating to health and longevity of herbivores, however, which are by their physiology more suited to diets high in roughage (Zollitsch *et al.*, 2004). A high cereal diet can also result in milk and meat production that is reliant on concentrates grown on arable land with high inputs of nitrogen fertiliser (Niggli *et al.*, 2009) and deforestation overseas in the case of imported soya and maize (El-Hage Scialabba and Müller-Lindenlauf, 2010). Novak and Fiorelli (2009) also highlight that replacing roughage by concentrates contradicts the European environmental policy to promote extensive use of maintained grasslands, which store significant amounts of carbon in soil (Freibauer *et al.*, 2004).

There are also some claims that diets high in tannins (e.g. diets with a high clover/legume content) produce less methane than grass-only diets through a suppression of fibre degradation in the rumen (Hess *et al.*, 2006). For instance, McCaughey *et al*. (1999) found that CH_4 emissions were reduced by 25% in the case of beef cows fed on alfalfa-based pastures, compared to grass only pastures. Although ruminant livestock farmers in the UK have been reluctant to grow alfalfa, due to a perception that forage legumes are difficult to grow and ensile, the situation is changing due to increased availability of disease resistant varieties and a growing awareness of the crop's usefulness as a high-yielding source of protein (Cotswolds Seeds, 2017, AHDB Beef & Lamb, 2016). Lampkin (2007) highlights that the use of such legumes could represent a potential "triple-gain" for climate change: reducing methane emissions from animals, building SOM and reducing reliance on synthetic nitrogen fertilizers. The increasing popularity of legumes on non-organic farms may also result in savings for the agriculture sector overall, although a recent meta-analysis, conducted as part of the UK Greenhouse Gas Platform project[7], concluded that there is insufficient evidence for high-legume/high-tannin pastures to be considered an effective option for lowering direct CH_4 emissions from ruminant livestock (McBride *et al.*, 2012).

3.5.4 Selection of effective greenhouse gas mitigation measures for agriculture

Due to the present uncertainties in estimating greenhouse gas emissions from agricultural practices (Smith *et al.*, 2007), the selection of effective mitigation measures by policy makers and industry is a difficult task. In particular uncertainties in this field relate to N_2O emissions, which can be greatly affected by local soil and weather conditions (Mathieu *et al.*, 2006). CH_4 emission estimates are also influenced by a range of factors such as feed intake and ruminal microflora (Johnson and Johnson, 1995). Climate and global change will also affect the future of agriculture, and the efficacy of mitigation options. For example it has been demonstrated that elevated atmospheric CO_2 concentrations could increase crop yields by 10-15%, which would reduce the demand for arable lands and non-renewable resource use per tonne of product (Smith *et al.*, 2007).

Novak and Fiorelli (2009) also highlight that any assessment of mitigation options should carefully consider the balance between their adverse and beneficial effects, with particular regard to practices diminishing productivity that may induce land-use change elsewhere in the world (Smith *et al.*, 2007) or the fact that hedgerows used to store carbon in vegetation can have many other valuable functions such as creating a windbreak or improving biodiversity (Jose, 2009).

Despite such difficulties, a number of studies have attempted to assess the combined-GHG mitigation potential of organic farming systems (i.e. including CO_2, CH_4 and N_2O) using Life Cycle Assessment (LCA) approaches, which

[7] www.ghgplatform.org.uk

offer a comprehensive and relatively consistent framework for assessing the environmental burdens of production processes (Notarnicola *et al.*, 2017, see Section 3.8.1 for a description of the LCA approach).

A comprehensive literature review comparing the Global Warming Potential (GWP) estimates from a range of LCA-based studies found no significant differences overall between organic and non-organic systems (Figure 3.5, Knudsen *et al.*, 2011). The authors note that lower yields in organic systems tended to offset the reduced use of manufactured inputs, leading to small differences overall when greenhouse impacts were expressed on a unit of product basis. Conversely, and in common with a more recent work in this area, organic systems performed much better than non-organic when GWP estimates were expressed per unit of land area (Knudsen *et al.*, 2011; Meier *et al.*, 2015).

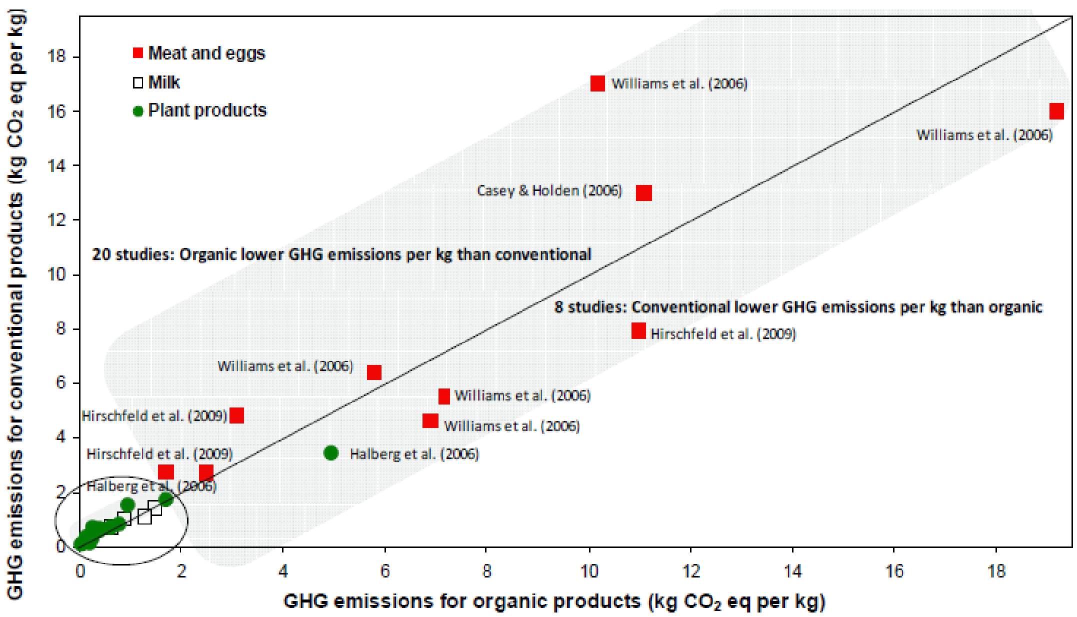

Figure 3.5: Literature review of conventional and organic products. Organic performs better above the line, worse below the line. Source: Knudsen *et al.* (2011)

The study by Knudsen *et al.* (2011) highlighted the importance of diet: beef had the highest greenhouse gas impact followed by lamb, pork, poultry and eggs.

The reviewers therefore note that the replacement of meat by plant products will have a greater impact than replacing conventional products with organic ones. Similar conclusions were drawn by Clark and Tilman (2017) who also found that organic systems result in similar GHG emissions per unit of product, and that shifting consumption patterns towards low impact foods would have a much greater impact on GHG reduction than a change in farm management practices.

Knudsen *et al.* (2011) also cite a study by a Danish supermarket, which found that consumers who buy organic products tend to eat less meat than those who do not (FDB, 2010), suggesting that the diets of typical organic consumers may be inherently more sustainable than those of the general population. The authors of the study also note that most LCA-based comparisons do not address the GHG offset that could be achieved through soil carbon sequestration within organic farming, and that the inclusion of this aspect may make the difference between the systems much clearer (Knudsen *et al.*, 2011).

3.6 Carbon sequestration in organic farming

Maintaining and increasing soil organic carbon (SOC) in agricultural systems is the mitigation option with the greatest potential (Smith *et al.*, 2008) and building reserves of soil carbon can increase the potential productivity of soil (Smith *et al.*, 2007). Practices that have been shown to increase SOM, such as the use of organic fertilisers and fertility building leys with legumes, are commonly found on organic farms (Smith *et al.*, 2011) and a range of long-term field trials have found higher organic matter contents in organically managed soils (Clark *et al.*, 1998; Loes and Øgaard, 1997; Mäder *et al.*, 2002; Mäder *et al.*, 2006; Marriott and Wander, 2006; Nguyen *et al.*, 1995; Reganold *et al.*, 1993). In addition to storing carbon, higher levels of SOM can enhance the nutrient buffering capacity, water holding capacity and microbial activity within soils and help to increase the soil's fertility (El-Hage Scialabba and Müller-Lindenlauf, 2010). Stockdale *et al.* (2001) also point out that the size of the active fungal population is often increased under organic management and increased microbial populations have been observed within organic farming in a long-term trial (Fließbach *et al.*, 2007).

A review of 32 peer-reviewed publications explored SOC contents within organic farming, revealing a 2.2% average annual increase in SOC within organic systems (Leifeld and Fuhrer, 2010) however, the authors highlighted that in 74% of cases the amount of organic fertiliser (e.g. manure and/or compost) in the organic systems exceeded that applied in the conventional. Leifeld and Fuhrer (2010) state that a truly unbiased comparison should be based on the same or similar crop rotation (i.e. including fertility building ley periods in both organic and non-organic systems) and the same rates of organic fertiliser application, as neither of these practices are exclusively found in organic farming. Although this suggestion is valid, a comparison of this kind would ignore the fact that organic farmers are more likely to be using a fertility building period and manures as a result of organic principles and standards. A review of field studies carried out in the US also found that legume based and manure and legume based organic management resulted in similar levels of SOM increase (Marriott and Wander, 2006), suggesting that the ley period alone is more significant than additions of manure, in terms of building soil carbon. Soussana *et al.* (2007) also found that temporary leys have the potential to contribute to C storage when net-exchange of carbon is considered on a site-by-site basis.

A more-recent study carried out by the Swiss Research Institute of Organic Agriculture (FiBL) reviewed 74 studies to identify differences in topsoil organic carbon under organic and non-organic management. The results revealed significantly higher SOC concentrations (+0.18±0.06% points), stocks (3.50 ±1.08 Mg C ha^{-1}) and sequestration rates (0.45 ± 0.21 Mg C ha^{-1} y^{-1}) within organic systems (mean values ± 95% confidence interval). The main reason for the difference was found to be the use of ley/arable rotations and the application of organic fertilisers (Gattinger *et al.*, 2012). When Gattinger *et al.* (2012) limited the assessment to organic farms receiving zero net-inputs, significant, positive differences in SOC concentrations and stocks were still found (0.13 ± 0.09% points and 2.16 ± 1.65 Mg C ha^{-1}, respectively) although the difference for sequestration rates was no-longer statistically significant (0.27 ± 0.37 Mg C ha^{-1} y^{-1}).

Although all of the studies assessed within the FiBL meta-analysis were based on pair-wise comparisons, and consisted of a mix of plot experiments, field trials and farm assessments, the authors note that many of the papers reviewed suffer from shortcomings that reduce their scientific value. In particular, many of the studies contained no data on baseline SOC concentrations which made it impossible to determine if the differences were due to the management itself, or the residual effect of the previous land-management. The authors of the study also point out that there are major issues relating to the shallow soil sampling depth (a median of 22.5cm across the studies). With over 50% of soil carbon residing in the subsoil (Batjes, 1996) this is an important omission.

It is also important to consider that soil carbon sequestered in arable soils is impermanent and is lost more rapidly than it accumulates (Freibauer *et al.*, 2004; Soussana *et al.*, 2004) and that even a single ploughing can greatly increase the rate of mineralisation and subsequent release of CO_2 (Reicosky *et al.*, 1999). To be effective, therefore, a conversion from arable to grassland should remain permanent (Novak and Fiorelli, 2009) although smaller increases in levels of SOC have been observed in some ley-arable systems, compared to continuous cropping (Johnston *et al.*, 2009). The total amount of carbon that can be sequestered following the adoption of organic farming will also depend on the SOM content at the time of conversion and the system type used as a comparator (e.g. whether both organic/non-organic systems are following a ley arable rotation, or whether the non-organic system is 100% arable, Powlson *et al.* (2011). Moreover, the addition of organic material to the soil can only be considered to provide a net benefit with regard to soil carbon if the application avoids a deleterious alternative. For instance, the application of animal manures and straw to arable soils is often associated with a soil carbon increase, however in a UK context virtually all of the manure and straw produced is already applied to soils, or incorporated soon after harvest (Powlson *et al.* 2011).

Long-term studies have also revealed that the rate of SOC accumulation in agricultural soils is non-linear and will often reduce over time as a new steady-state is reached. For instance the 140-year Broadbalk Experiment at Rothamsted Research, UK, found that on the 'farmyard manure' plots the rate of increase was greatest in the early years of application, and reduced over-time as the soil approached a new state of equilibrium (Powlson *et al.*, 2011). It is therefore essential that any initial, large gains in soil carbon sequestration, resulting from the addition of manures or other organic materials, are not extrapolated year on year under the assumption that the same increase will continue to occur indefinitely (Powlson *et al.*, 2011). Smith *et al.* (2007) also highlight that terrestrial biomass sources only remove carbon from the atmosphere until the maximum capacity for the ecosystem is reached – this phenomenon is referred to as 'sink saturation' and is shown in Figure 3.6.

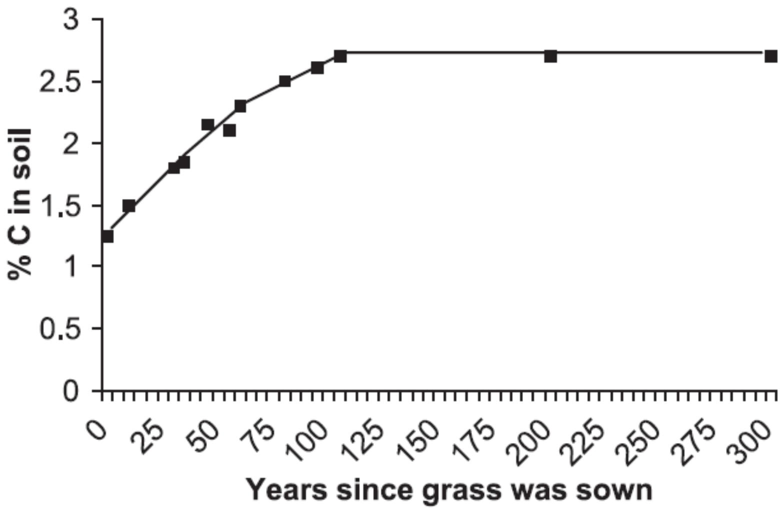

Figure 3.6: The accumulation of total soil carbon in silty clay loam soils at Rothamsted, UK, when old arable land is sown to permanent grass. Adapted from nitrogen content in Figure 18.10 of Jenkinson (1988). Source: Freibauer *et al.* 2004.

Reduced tillage also has the potential to increase rates of soil carbon sequestration (King *et al.*, 2004) although no-tillage is difficult in organic agricultural systems because the associated development of weeds cannot be controlled with the use of herbicides, only by mechanical weed control (El-Hage

Scialabba and Müller-Lindenlauf, 2010). Recent work has also highlighted that increases of SOC resulting from reduced tillage are often associated with higher concentrations near the soil surface, rather than an increase in SOC stocks, and that the need to combine reduced tillage systems with inversion tillage to control perennial weeds will almost certainly result in any gains being lost through mineralisation (Powlson *et al.*, 2012). The same study highlights that there is little room for the expansion of reduced tillage in the UK, and that only certain soil types are suited to this method of cultivation. Other authors have highlighted that the larger soil aggregates, low gas diffusivity and greater water retention near to the soil surface, that results from reduced tillage, can make the soil less aerobic and lead to higher N_2O emissions (Ball *et al.*, 2008; Ball *et al.*, 1999)

Despite uncertainty over the rates of SOC increase resulting from organic practices on agricultural land, previous studies have highlighted the improvement in other soil properties that can result from an adoption of organic management. For example Siegrist *et al.* (1998) found that the adoption of organic agriculture can significantly reduce soil erodibility on a silty loam soil, in addition to increasing earthworm biomass and density. Reganold *et al*. (1987) also found significantly higher moisture contents on soils under organic management (attributed to high SOM levels) and reported that levels of water erosion were nearly four times greater on the non-organic land. Reganold *et al.* (1987) state that the difference in erosion rates between organic and conventional farms is related to the different crop rotation systems- highlighting that only the organic farm in their comparison included a green manure crop every third year and that the organically managed land had fewer cultivations. It is therefore suggested that organically managed soils are less prone to the effects of extreme weather events such as flooding, water-logging and drought (El-Hage Scialabba and Müller-Lindenlauf, 2010). Organically managed land also tends to be more diverse, with a greater proportion of the land-area devoted to features which improve biodiversity and encourage positive interactions. The presence of these features is encouraged by the IFOAM basic standards, which state that organic producers “shall take measures to maintain and improve landscape and enhance biodiversity" (IFOAM, 2002). Agroforestry systems have similar effects and their uptake is also encouraged within different standards for organic agriculture (IFOAM, 2002).

3.7 Previous attempts to explore the impacts of 'up-scaling organic'

Several studies explored the impacts of a large-scale uptake of organic agriculture. The methods applied and results derived are summarised in the following sections.

3.7.1 Up-scaling studies – UK based assessments

The most recent study exploring the impacts of a 100% conversion in a UK context was completed by the University of Reading's Centre for Agricultural Strategy. Jones and Crane (2009) considered three separate approaches to exploring how much food could be produced if all agriculture in England and Wales were organic:

- **Weighting by farm type:** In this approach, the current mix of organic farm types was assumed to change to become consistent with the current mix of conventional farm types at a national level.
- **Weighting by yield:** in this approach organic yield ratios were applied to the conventional production figures (e.g. a reduction of 10-50% depending on the commodity), which avoided the issue of farm numbers or farm systems.
- **Supply side modelling involving the proliferation of model organic farms in the colonization of available agricultural land:** this approach was not applied due to time constraints, however the methodology that could have been applied with more resource was outlined in Jones and Crane (2009). For this modelling approach, a linear programming-based model would be constructed and tasked with meeting the demand for domestically produced commodities using the land base within a particular region. The model would then populate a given region with organic farm types required to meet the specified

demand. The authors state that this option would provide the most realistic results by allowing the organic systems to adapt in order to meet consumer demand across the various regions.

Jones and Crane (2009) found that when applying the weighting by farm type method, a conversion to organic production would increase the amount of 'minor cereals' (i.e. oats, rye, triticale) dramatically, exceeding conventional production by 2.2 million tonnes, whereas the production of wheat and oilseed rape would reduce, as these two crops are currently under-represented on organic farms in England and Wales (there is currently no domestic market for organically produced oilseed rape (Jones and Crane, 2009). When weighting by yield, for all cereals together, a wholly organic agriculture would produce approximately 59% of the conventional output. For livestock, the 'farm types' estimate found that the amount of beef that would be produced under organic management would exceed the amounts currently provided by conventional farming, due to an increased presence of suckler herds on arable farms, as a result of increased areas of grass/clover ley. Conversely within the 'by-yield' estimate the total amount of beef production was somewhat reduced due to lower stocking rates on most organic farms. Pigs and poultry would also reduce because of much lower stocking densities. The projections for organic milk production ranged from 61-70% depending on whether the converting dairy farms retained their original size (resulting in the lower estimate) or increased their production intensity.

Table 3.2: Estimates of organic production as a proportion of conventional production derived from two raising approaches (Jones and Crane, 2009)

	Organic output as a share of conventional production (%)	
	Weighting by organic yields	Weighting by farm type
Wheat	67.7	33.4
Barley	73.6	47.5
Oats	42.2	415.4
Mixed cereals	71.8	420.6
Oilseed rape	0.0	4.0
Peas/beans (stock)	83.3	98.0
Potatoes	76.9	142.6
Sugar beet	109.2	47.6
Milk	70.2	61.1
Beef (total head)	70.9	168.5
Steers, heifers, young bulls	75.3	189.8
Calves	2.8	8.3
Cows, adult bulls	62.1	95.5
Sheep	87.5	155.4
Pigs	N.A.	27.2
Poultry meat (birds)	N.A.	29.7
Eggs (dozen)	N.A.	73.3

The authors also note that a wholly organic agriculture would lead to very large savings in in-organic agrochemical use, although volumes of other inputs, in particular labour, would rise. Benefits for the wider rural economy in terms of employment were therefore reported as being self-evident, although the authors note that the high labour requirements may be perceived as a barrier to conversion in some cases, especially for arable farms which may be operated largely by a sole worker (i.e. the farmer).

A major difficulty highlighted within the University of Reading study is that scaling on the basis of farm-type assumes that current demand patterns for organic products will remain the same (i.e. there would be more emphasis on horticultural crops and less emphasis on crops for processing such as oilseeds and sugar beet). Conversely, the approach of applying organic yield ratios to the area of organic crops ignores the significance of typical organic rotations and the present dominance of mixed farming systems within organic agriculture. In view of this Jones and Crane (2009) state that most reliance should be placed on estimates when the values obtained through the two approaches concur, as is the case for peas and beans in Table 3.2.

An earlier study completed by Lampkin (1994) also explored the effects of a widespread conversion to organic agriculture in the UK. To implement the organic scenario a range of crop area and yield adjustments were applied to the conventional production data, as follows.

- A reduction in the area of cereals, by a maximum of 25%
- An expansion in the area of oats, at the expense of wheat and barley (a characteristic common to organic farms)
- An expansion in the area of potatoes, vegetables and fodder crops
- A reduction in sugar beet and oilseed rape areas due to difficulties associated with the organic production of these crops
- An expansion in grain legume crop areas, particularly because of their usefulness in organic rotations in terms of providing nitrogen through biological fixation and because of their flexibility (i.e. can be used as either livestock feed or for human consumption)
- Grass ley areas were expanded, particularly in arable areas
- The area of permanent pasture and rough grazing was reduced slightly, due to increased forestry and environmental conservation activities which would lead to a general increase in woodland and 'other land' areas and because long-term grass leys would be brought back into cropping rotations

The yield output assumptions were specified as ratios (organic:conventional) and combined with the crop area assumptions to define total outputs at national level. The yield percentages ranged from 60% in the case of organic wheat to 80% in the case of grain legumes. Dairy livestock numbers were estimated to remain the same as conventional under organic management although milk yield per cow (and total output) was reduced by 10% as the result of a switch to diets low in concentrate. Stocking rates were predicted to fall by 20% in the case of grazing livestock and 40% in the case of pigs and poultry, illustrating the lower stocking rates and reduced feed availability under a 100% organic scenario.

The study found that the UK's self-sufficiency under organic management would reduce to a similar level to the 1970s, although the author highlights that levels of food self-sufficiency in the UK may have been overestimated due to a reliance on livestock feed imports (Lampkin, 1994). The study emphasised that the assumptions made are necessarily speculative, but provide a useful starting point on which to base further work (Lampkin, 1994).

3.7.2 Up-scaling studies – other EU countries

A different approach to 'scaling-up-organic' was used by Zerger and Bossel (1994) in which a range of prototype farms were defined for 15 'production regions' in Germany, and 75 'state indicators' defined for each region over each point in time (every five years). These indicators included the national food demand, for which three scenarios were explored i.e.

1. Maintenance of 1983 per capita meat consumption
2. Increasing per capita meat consumption
3. Reduction in meat consumption to 1960 levels

Three productions scenarios were also explored within the study, i.e.

1. Intensification
2. Business as usual
3. Ecologization (organic farming)

The organic production scenario was found to result in much lower levels of grain production (approximately 40% lower than "business as usual") due to a reduced area and production intensity. There would also be a decrease in the area of sugar beet and an increase in potato production. In contrast to Jones and Crane (2009) cattle numbers were also reduced by about 20%. Interestingly the study raised a major question of whether a large scale conversion to organic farming could be financed as the organic option had the lowest profit over a 40 year period compared to the "business as usual" and "intensification" options. The authors therefore highlight that this option can only be pursued if risk reduction measures and an appropriate policy environment are put in place. However, this study assumed conventional prices for all products sold; the economic situation would clearly change if organic premiums were applied.

A team of researchers in Denmark also investigated the impacts of a restructuring of agriculture toward organic approaches over a 30 year time period (The Bichel Committee, 1999). Typical farm-structures and national organic yield data were combined within an upscaling approach to derive estimates of production and crop area at a national level (Ørum *et al.*, 2000). Two yield levels were explored within the analysis, i.e. the “present yield level” based on current organic practice, and an “improved yield level” in which it was assumed that organic cereal production could be increased by 15% and clover grass production by 10%. Two different levels of livestock feed imports to Denmark were also used in the organic scenarios, i.e. 1. No feed import, complete self-sufficiency in feed, and 2. 15% imported feed for ruminants and 25% for non-ruminants, with imported feed amounts determined on the basis of feed-energy intake.

The substantial reduction in output under organic management for the major crop and livestock types is illustrated in Table 3.3 The report illustrated that a considerably lower rate of production for pigs would have serious consequences for employment in primary production and processing, and that a number of supply industries would experience a fall in the demand for goods and services. The Gross National Product (GNP) would therefore be reduced by 1-3%, corresponding to an annual reduction of DKK 11-26bn (£ 1.3-3.1 million, exchange rate on 23/06/2017). Private consumption would be reduced by 2-5%, corresponding between DKK 1,900 and DKK 4,700 per capita per year (equivalent to £ 228-564) due to a reduction in agricultural wages, as a result of the reduced output (The Bichel Committee, 1999).

Table 3.3: Changes (%) in production under organic scenarios investigated by the Bichel Committee (Jacobsen and Fransden, 1999)

	Present level of yield		*Improved level of yield*	
	0-import	*15-25% import*	*0-import*	*15-25% import*
Cereal	-62.0	-53.8	-52.9	-44.7
Rape	-3.2	-100.0	-11.6	-100.0
Potatoes	-79.8	-79.8	-79.8	-79.8
Sugar beet	-54.4	-54.4	-54.4	-54.4
Greenfeed	57.6	53.7	66.0	63.4
Dairy farms	0.0	0.0	0.0	0.0
Pigs and poultry	-69.1	-29.2	-54.3	-7.2
Total	*-33.8*	*-20.4*	*-26.2*	*-10.1*

In addition to calculating the effect on yield, a sub-group within the project calculated the environmental impacts of a 100% organic scenario (Hansen *et al.*, 2001). This group found a much lower surplus of N within the organic farming scenarios (146–245 million kg N yr^{-1}) compared to that of current Danish farming methods (418 million kg N yr^{-1}). Calculations within this sub-group also demonstrated that a switch to organic production could result in a reduction of 9 to 53% in net energy consumption, depending on the amount of imported feed. Greenhouse gas emissions were predicted to reduce by between 13 and 38%, mainly as a result of the reduced nitrogen inputs and a reduction in animal numbers (The Bichel Committee, 1999).

3.7.3 Up-scaling studies – global assessments

Halberg *et al*. (2006) also used a modelling approach to explore the implications of a global conversion to organic agriculture in terms of food security. This study used the IMPACT model (see description in section 3.8.2 below) which offers a method for analysing baseline and alternative scenarios of food demand, trade income and population. Through a review of the available literature, the authors found that organic agriculture results in yields that are 15-35% lower than conventional, and that yields would be lower when crop failures and the need for a fertility building ley in the rotation were accounted for. Despite this reduction, the study found that a large-scale conversion to organic farming would not result in negative effects on global food availability, compared with a baseline of food production over a 30 year period from 1995 onwards, especially if combined with food security policy initiatives. The authors also highlight that there may be other complementary advantages of supporting organic farming in Europe, such as decreasing overproduction, reducing environmental impacts and encouraging the development of a more 'multi-functional agriculture' (Halberg *et al.*, 2006). There was even found to be a positive effect on food security in Sub-Saharan Africa and the study concluded that the partial introduction of modern organic farming systems in this region could result in yield increases of at least 50%.

Badgley *et al.* (2007) also explored the implications of a 100% conversion to organic production on global food supply, through an application of organic yield ratios to FAO-derived statistics on food production within ten categories covering the major plant and animal components of human diets. The yield ratios were obtained through a review of 293 comparisons, which were mainly derived from peer-reviewed scientific literature, although some yield data from conference proceedings, websites and technical reports were included. The average yield ratios for the 10 groups are shown in Table 3.4 below. The overall average yield ratio for all crop types was found to be 1.32 (Organic : conventional) at a global level. Total N supply was estimated by multiplying the area currently in crop production by the average amount of N available from legumes established during winter fallow or between crops. The total N supplied by leguminous crops was estimated to be 140 million Mg, "which is 58 million Mg greater than the amount of synthetic N currently in use". The authors therefore suggest that rates of biologically fixed N and release can match N availability with crop uptake and achieve yields equivalent to high yielding conventional crops.

Table 3.4: Average yield ratio (organic: non-organic) and standard error (S.E.) for ten individual food categories recognized by the FAO. *N* = number of comparisons for each crop, Av. = average yield ratio based on the results from the comparisons

	(A) World			(B) Developed countries			(C) Developing countries		
Food category	*N*	Av.	S.E.	*N*	Av.	S.E.	*N*	Av.	S.E.
Grain products	171	1.312	0.06	69	0.928	0.02	102	1.573	0.09
Starchy roots	25	1.686	0.27	14	0.891	0.04	11	2.697	0.46
Sugars and sweeteners	2	1.005	0.02	2	1.005	0.02			
Legumes (pulses)	9	1.522	0.55	7	0.816	0.07	2	3.995	1.68
Oil crops and veg. oils	15	1.078	0.07	13	0.991	0.05	2	1.645	0.00
Vegetables	37	1.064	0.10	31	0.876	0.03	6	2.038	0.44
Fruits, excl. wine	7	2.080	0.43	2	0.955	0.04	5	2.530	0.46
All plant foods	266	1.325	0.05	138	0.914	0.02	128	1.736	0.09
Meat and offal	8	0.988	0.03	8	0.988	0.03			
Milk, excl. butter	18	1.434	0.24	13	0.949	0.04	5	2.694	0.57
Eggs	1	1.060		1	1.060				
All animal foods	27	1.288	0.16	22	0.968	0.02	5	2.694	0.57
All plant and animal foods	293	1.321	0.05	160	0.922	0.01	133	1.802	0.09

If no data were available for individual crop types (e.g. tree nuts), then Badgley *et al.* (2007) used the average yield ratio for all plant foods or animal foods where relevant. The results suggest that organic production has the potential to provide a substantially greater (c50%) amount of food than is currently produced, leading the authors to conclude that “organic production has the potential to support a substantially larger human population than currently exists”. In common with Halberg *et al.* (2006), the study found high yield ratios for farms under organic management in the developing world (1.80 organic: conventional), implying that food security could be increased in these areas. Although the authors highlight that, at present, agriculture in developing countries is generally less intensive than in the developed world, and that organic agriculture is often compared with local, resource-poor methods of subsistence agriculture (Badgley *et al.*, 2007).

Connor (2008) criticises the outcomes of the Badgley *et al.* (2007) study, pointing out that the authors fail to realise that such a large increase in organic production would result in competition for limited organic nutrients and that crop yields and cropped areas will fall as an increasing proportion of land is devoted to fertility building ley periods. Connor (2008) also points out that the assumption by Badgley *et al.* (2007) that 100% of arable land could accept an additional legume crop is misleading and that for the multiple cropping systems of south-east Asia and China, organic agriculture could not possibly provide the quantity of N required per hectare (300-400kg) for producing the combined grain yields of >10 tonnes/ha.

A more recent study by de Ponti *et al.* (2012) found that of the total 293 yield data entries used by Badgley *et al.* (2007) only 14% (42) met their quality criteria (i.e. relating to yield data being outdated, unrepresentative yield levels, insufficient information on treatments). Within their meta study, de Ponti *et al.* (2012) found that on average organic yields were 80% of those obtained under conventional agriculture, with a standard deviation of 21%. The relative yields differed greatly across the regions of the world; in Northern Europe the relative yield was lowest (70%) and highest in Asia (89%). The differences found for individual crop groups are summarised in Table 3.5 below:

Table 3.5: Averages and ranges for the organic-conventional relative yields of selected crop groups and crops. Source: de Ponti *et al.* (2012)

Crop type	Relative yield average %	Range %
Cereals	79	40-145
Root and tuber crops	74	37-114
Oilseed crops	74	41-114
Vegetables	80	21-140
Fruits	72	20-94
Apple	69	44-92
Other food crops	92	78-106
Fodder crops	86	42-177

The authors of the de Ponti *et al.* (2012) study state that the comparatively low yield gap between organic and conventional in the tropics can be accounted for by the fact that crops commonly grown in these regions, e.g. soybeans and other-pulses, have the ability to fix nitrogen and therefore require little/less artificial N-fertiliser.

3.8 Modelling approaches for assessing a change in agricultural land use– an overview and inter-comparison

The use and development of agricultural models has increased rapidly over the last 30 years (Brockmeier and Urban, 2008) with the use of such approaches being considered an effective means to determine and assess complex relationships as an aid to decision making (Wang *et al.*, 2009). In the beginning models were very limited in scope whereas today the use of global and/or regional multi-impact models covering a range of environmental, economic and social indicators is commonplace within the field of policy assessment and international trade negotiations (Brockmeier and Urban, 2008).

The wide variety in modelling approaches used for assessing the environmental impact of farming systems is illustrated by Payraudeau and van der Werf (2005) who carried out an analysis of six methods: 1. Environmental Risk Mapping, 2. Life Cycle Analysis, 3. Environmental Impact Assessment, 4. Multi-agent Systems, 5. Linear Programming and 6. Agri-environmental indicators. Payraudeau and van der Werf (2005) highlight that the models can be split into those that are input related, e.g. considering the amount of fertiliser used, and effect related, e.g. considering the incidence of soil erosion. Payraudeau and van der Werf (2005) also state that effect based indicators are preferred, as they allow a more direct assessment of environmental impact. Models also vary greatly in their detail, some using a time intensive approach to provide a comprehensive assessment of a limited range of indicators (e.g. Life Cycle Assessment-based models) whilst others provide a rapid overview of farming system performance against a broad range of environmental, economic and social sustainability criteria (Schader *et al.*, 2014).

For this study, the following models were considered in terms of their potential application in exploring the production and environmental impacts of a 100% conversion scenario:

- The Cranfield Agri-LCA: a suite of input and effect based models developed within a Life Cycle Assessment (LCA) framework
- The Silsoe Whole Farm Model: an input and effect based Linear Programming model
- The Land Use Allocation Model (LUAM): an input based-static Linear Programming model that represents the agricultural system in England and Wales as if it were a single farm
- Farm Modelling Information System – FARMIS: an input based Positive Mathematical Programming Model that includes agro-environmental indicators
- International Model for Policy Analysis of Agricultural Commodities and Trade (IMPACT). A commodity market model for developing economic scenarios in relation to food security and water use
- Common Agricultural Policy Regionalised Impact Analysis model (CAPRI) a commodity market model for assessing environmental and economic implications (food supply and trade) of CAP reform

An overview of each model and its features follows.

3.8.1 Agri-LCA model

The Agri-LCA model, developed at Cranfield University (Williams *et al.*, 2006) is a farm-systems based model developed to analyse and compare the environmental impacts of major agricultural commodities. The Life Cycle Assessment (LCA) approach used within the model can be defined as a systematic process to evaluate the environmental burdens associated with the production and/or delivery of a product or activity. Within agriculture, LCA is used to calculate the burdens associated with one unit of a food commodity, e.g. 1kg of wheat or meat, defined as the 'functional unit' (Williams and Audsley, 2008). Besides the functional unit there can be co-products or waste items such as straw, together with emissions to the environment, for example nitrate (NO_3-) to water and nitrous oxide to air (N_2O), as illustrated in Figure 3.7. Inputs to the system are usually traced back to the primary resource, e.g. the coal or uranium used to generate electricity or the energy required to produce steel, plastic and other materials required for the manufacture of tractors (Williams *et al.*, 2006).

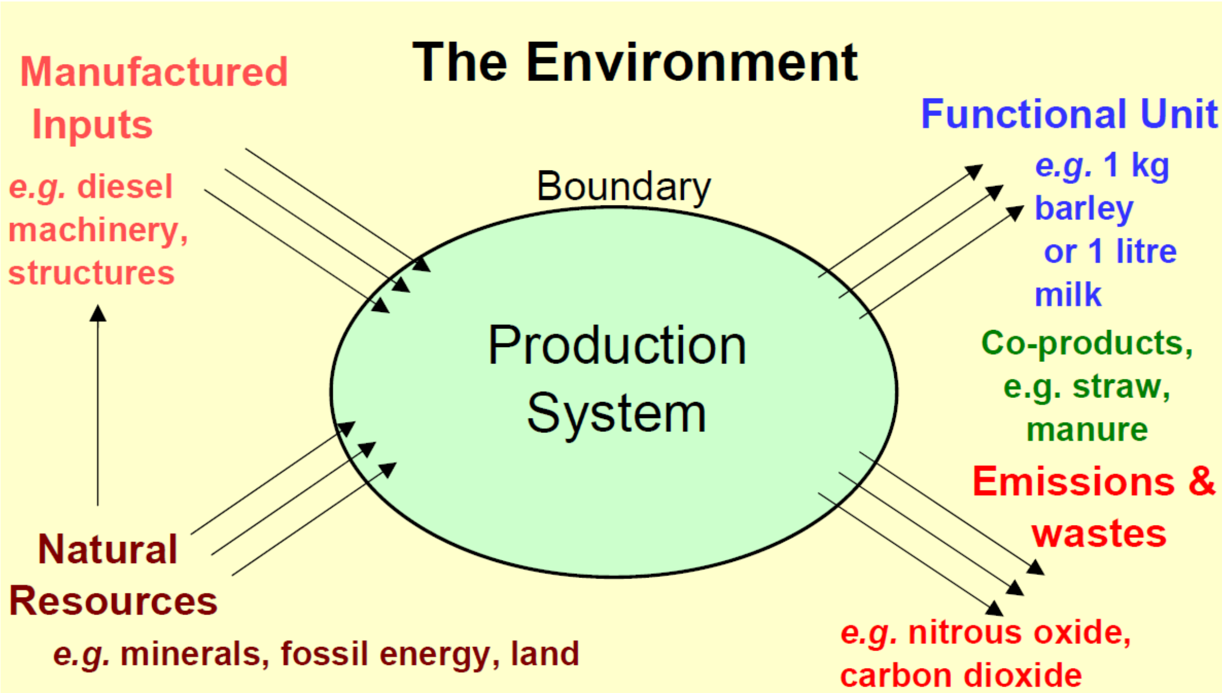

Figure 3.7: Outline of Agricultural LCA process. Source: Williams *et al.* (2010)

The LCA approach used within the Cranfield model is enshrined in the International Standards ISO14040 and 14044 (British Standards Institute, 1997, 2006). According to these standards the four stages of an LCA are:

1. Goal and scope definition
2. Life Cycle Inventory Compilation
3. Life Cycle Impact Assessment
4. Interpretation

Phase 1 refers to the definition of the functional unit and system boundaries, with stage two referring to the gathering of information on relevant energy and material inputs and outputs (LCI). An evaluation of the potential environmental impacts is then made by using quantitative characterisation indicators based on an Environmental Model (Life Cycle Impact Assessment - LCIA, Renó *et al.*,

2011) before the results are interpreted and conclusions drawn. Steps 1-4 are not always sequential, as it may be necessary to re-define the goal and scope of the study in light of missing or additional data, identified in stage 2 for example.

Williams *et al*. (2006) highlight that agriculture has particular features that are not relevant to the LCA of industrial processes, relating to the fact that the effect of farming systems must be considered in the long-term, with particular regard to major nutrients, soil carbon and weed accumulation. The consequences in these areas must be examined through process modelling to ensure that the system being modelled does not result in the depletion of soil reserves or significant accumulation (e.g. of nitrogen) over time. Unlike industrial products, or individual arable crops, livestock also pose certain difficulties. It is not possible to consider an animal product in isolation, as genetic flows, varying nutritional requirements according to the metabolic stage of the animal(s) and the production and subsequent application of manures all have to be considered. The Agri-LCA models allow for the consideration and adjustment of these processes through farm-system modelling approaches including process models applied to soils, crops, post-harvest activities and animal production. Nutrient flow and breeding/replacement models are also applied to the livestock and arable sectors (Williams and Audsley, 2008).

The indicators assessed through the Agri-LCA models include Eutrophication Potential (EP) measured in terms of phosphate (PO_4^{3-}) equivalents, the acidification potential, quantified in SO_2 equivalents (1kg NH_3-N is equivalent to 2.3kg SO_2). Primary energy use is also recorded in Mega Joules (MJ, i.e. 10^6 Joules). As can be seen in Figure 3.8, a figure is also given for Abiotic equivalent, in kg. This refers to Abiotic Resource Use (ARU), i.e. a use of natural resources aggregated using the method of the Institute of Environmental Sciences (CML) at Leiden University (Guinée, 1995). This method puts many elements and natural resources on a common scale, related to their scarcity. It is quantified in terms of the mass of the element antimony (Sb), which was an arbitrary choice (Williams *et al.*, 2006). Environmental burdens are assessed in terms of their impact over an extended time-period (e.g. Global Warming Potential values are expressed over 20, 100 and 500-year time frames). The models do not therefore simply provide a snapshot of current practice but an overview of the longer-term impacts associated with contrasting agricultural management approaches.

Export Main Results to csv File (Unique file name generated automatically) File is readable by Excel			4	5	6	7	8	9
Commodity	Reference	Unit	Primary Energy used, MJ	GWP 500, kg CO2e	Eutrophication potential, kg PO4 equiv	Acidification potential, kg SO2 Equiv	Pesticides used, Dose ha	Abiotic ant equ, kg
Pig meat	1,000	kg dwt	22116	3167	29	90	2.8	21.6
Poultry	1,000	kg dwt	32724	3797	27	56	3.3	24.2
Beef	1,000	kg dwt	36389	7543	98	217	1.8	23.4
Sheep meat	1,000	kg dwt	23196	8365	101	87	1.2	12.9
Milk	1,000	litres	2725	539	4	9	0.2	2.3
Eggs	16,667	60 g eggs	18023	2222	23	60	2.4	11.8
Horse	500	kg horse	8429	1216	8	6	0.4	4.8

Figure 3.8: Sample of results from Agri-LCA model (www.agrilca.org)

The system boundary for the model is specified at the ‘farm gate’, i.e. burdens associated with the transport and subsequent distribution of products following their production on farm are not considered, although some on-farm processing such as grain drying, milk cooling and potato storage is included (Williams *et al.*, 2006). Product quality is also an important consideration within the model; for example meat is quantified as the edible carcass weight as used in statistics provided by the Meat and Livestock Commission (MLC), whereas milk is defined as the quantity of the fat-corrected product and oilseed is simply the amount harvested at a specified dry matter concentration (Williams *et al.*, 2006).

Allocation of burdens is achieved by economic value and by system expansion with regard to manure (i.e. the burdens associated with avoided manufactured N fertiliser are discounted from the crops fertilised with manure-N). Average yields are assumed within the Agri-LCA model for Grade 3a land (Bibby *et al.* 1969 in Williams *et al.* 2006). Yields can also be adjusted for other grades of land by using linear coefficients (from Moxey *et al*. (1995) in Williams *et al.*

2006) as illustrated in Table 3.6. Grassland yields are calculated using the grass site class system (Brockman and Gwynn, 1994). The data sources used within the model are established inventories and factors for industrial process for example the John Nix farm Management Pocketbook (2004-2005), Agro Business Consultants (2002-2005), Lampkin *et al*. (2002-2005), MLC yearbooks for pigs, sheep and beef, websites of organisations such as the Milk Development Council and Defra statistics. Emission factors for ammonia, nitrous oxide and methane are sourced from the national inventories which also supply activity data. Some values were also developed within the Williams *et al*. (2006) study, based on some of the data sources listed above, inputs from Audsley *et al.* (1997) and the Ecoinvent data source (provided within the SimaPro platform).

Table 3.6: Land grade factors used to scale yields within the Agri-LCA

Land Grade	Scaling Factor
2	0.88
3a	1.00
3b	1.08
4	1.12

Nitrogen losses are adjusted within the Agri-LCA using a range of non-organic and organic crop rotations that contain representative crops and linear-relationships based on simulations with the SUNDIAL model (Smith *et al.*, 1996) which are applied for nine combinations of soil type and rainfall (i.e. sand, clay, loam and high, medium and low rainfall). The SUNDIAL simulations were run for long enough to ensure that the modelled rotations reach a steady state, i.e. with the soil organic N content being the same at the start and end of a rotation. The SUNDIAL-derived N loss estimates are combined with land occupation estimates for each crop by soil/rain class to determine the total impact. Fossil energy use is also calculated for each crop as a function of soil type and tractor power as described in Williams *et al.* (2006).

The environmental influence of changes in animal diet, breed and management are assessed through changing the input values to the models (e.g. daily liveweight gain, annual fat-corrected milk yield, housing system) which in-turn affects metabolisable energy (ME) and crude protein (CP) requirements and N losses from manures and slurries (e.g. leaching, volatilisaton and denitrification). Compound feed composition data are also applied and can be adjusted to determine embedded impacts of feed production overseas. Direct CH_4 emissions are calculated as a function of liveweight gains, dry matter intake (adjusted in accordance with the forage component of the diet) and milk yields in the case of dairy cows.

The Cranfield Agri-LCA model was used in a study carried out on behalf of the Food Climate Research Network and World Wildlife Fund (WWF) to assess the potential to reduce greenhouse gas emissions from the UK food system by 70% from a 2005 baseline (Audsley *et al.*, 2009; Williams *et al.*, 2011). A total of 21 production and technical measures were analysed together with eight behavioural measures, mainly relating to dietary change. Modelling was used within the study to combine measures where this was possible, and reductions were considered beyond the farm gate, e.g. in processing, packing and distribution areas.

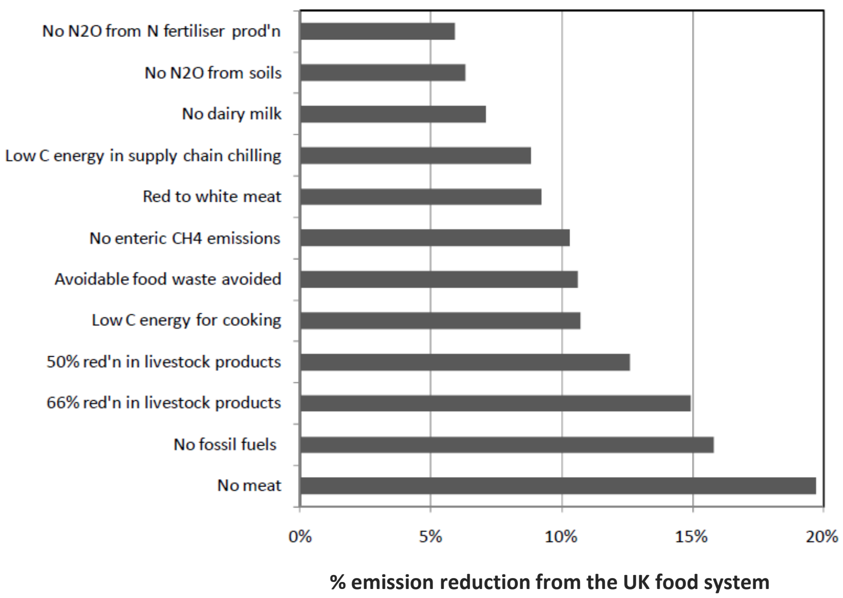

Figure 3.9: Effectiveness (% emission reduction) of the top 12 mitigation measures included within Williams *et al.* (2011)

No single measure or combination of measures was found to be capable of reducing emissions by more than about half. The single most effective measure within the study was ‘no meat’ with the technical measure of ‘no fossil fuels’ the next most effective (Figure 3.9) although it should be noted that all of the reductions assumed a 100% implementation which is unlikely for most of the measures considered.

3.8.2 IMPACT (International Model for Policy Analysis of Agricultural Commodities and Trade)

The IMPACT model was developed by the International Food Policy Research Institute (IFPRI) in Washington DC in the early 1990s. The policy-focussed model was developed to assist in the development of a long-term vision with regard to the actions that need to be taken to feed the world and reduce poverty whilst protecting the natural resource base (Rosegrant *et al.*, 2008). The modelling approach allows for an assessment of alternative scenarios encompassing global food supply and demand, population, trade and income (Halberg *et al.*, 2006). The model also contains regional sub-models which allows supply, demand and prices for agricultural commodities to be determined. Elasticities are incorporated within the supply and demand functions to approximate the underlying production and demand functions, with international markets being used to determine agricultural commodity prices (Rosegrant *et al.*, 2008).

The model covers a wide range of commodity groups and countries with trade between countries and regions being represented through a series of sub-models. Linear and non-linear equations are used within the model to estimate underlying production and demand functions. Prices are determined through annual world commodity prices, updated annually, with demand for agricultural products being a function of prices, income and population growth. Growth of crop production is also considered through the inclusion of a number of components, such as advancements in research and biotechnology, developments in infrastructure, education and the development of markets (Halberg *et al.*, 2006).

The model has been used in several important research publications to assess food demand and security at the national level. For example, the model was used by Msangi and Rosegrant (2009) to explore the impacts of several key drivers of change in food systems between 2007 and 2050, and possible entry points for policy intervention. The study found that an expansion in biofuel production, necessary to meet the USA's renewable fuel targets by 2022, would be accompanied by a net decrease in availability and access to food. As can be seen from Figure 3.10 an increase in the rate of biofuel expansion could have a detrimental effect on the number of malnourished children between 2000 and 2025.

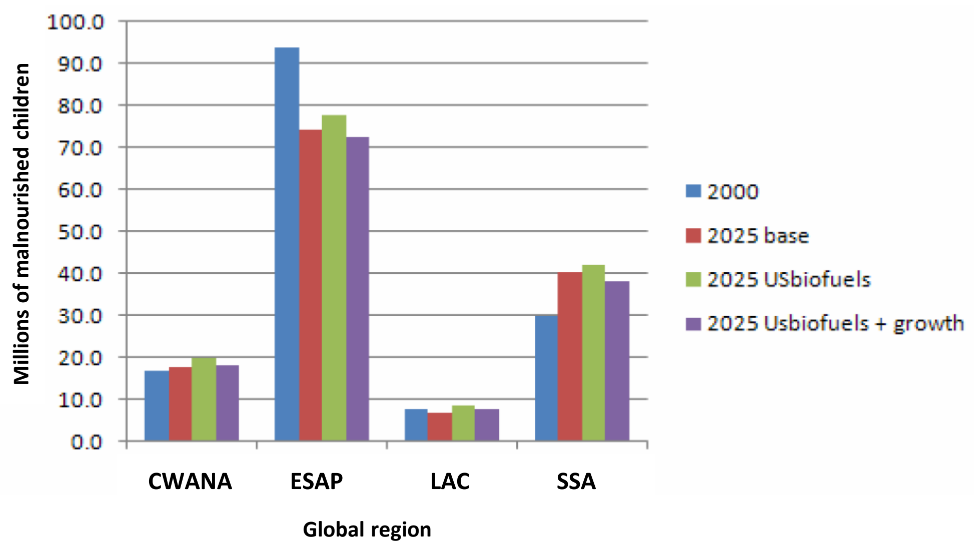

Figure 3.10: Trends in child malnutrition to 2025 under baseline case, biofuels and yield growth scenarios. CWANA = Central and West Asia and North Africa EASP = East Asia and Pacific, LAC=Latin America and the Caribbean, , SSA=Sub Saharan Africa (Msangi and Rosegrant, 2009)

The report therefore suggested that policy interventions should focus on avoiding the use of food crops in the production of biofuels (e.g. ethanol and biodiesel). The same report found that an increased investment in AKST (Agricultural Knowledge, Science and Technology) would result in greatly enhanced availability of calories, particularly in Sub-Saharan Africa and that the expected food price impacts of climate change would have strong implications for livestock production in particular, due to much higher costs of maize feed.

3.8.3 Silsoe Whole Farm Model

The Silsoe Whole Farm Model is a computer based multiple objective Linear Programming (LP) model developed for exploring a number of farming scenarios (Annetts and Audsley, 2002). The model aims to find the optimal solution from a range of cropping and machinery options for an individual farm with particular physical, economic and climatic characteristics. Annetts and Audsley (2002) highlight that linear programming is a useful method for decision making in the planning of production on arable farms in the UK, as it captures conflicts between a number of choices of enterprise, which each have different requirements with regard to time/labour inputs, variable costs (e.g. seeds, fertiliser) and the time available which will vary according to farm location, the time of year and the soil type.

The crop rotation, machinery use and cost of machines, cost of repairs, use of inputs (fertiliser, herbicides), fuel use and timing of operations are all included in the model together with the level of manpower available, subject to constraints. The climate and soil characteristics are the main physical constraints on the farm. Users of the model can choose to optimise long-term profit or a multiple objective of profit, risk and environmental criteria. The model optimises the weighted sum of component objective functions, which calculate the net profit and environmental outcome subject to the following constraints (Annetts and Audsley, 2002):

- Limits on the amount of machinery and time available for each period of the year
- Restrictions on the total amounts or total area of activities
- The requirement for crop operations to be sequenced including crop rotation sequencing
- Livestock grazing and feeding constraints

The model can therefore be used to examine the effect of changes on a farm such as alternative crop yields and new crops, livestock, machinery or cultivation techniques. The model can also assess the effect of changes in the price paid for crops sold or inputs used (e.g. the price of tractor diesel).

The effectiveness of the model was demonstrated by Audsley *et al.* (2006) who assessed the impact of future European agricultural land use scenarios, under various climate projections derived from the HadCM3 and PCM climate model outputs for 2001-2100. The results from the Silsoe Whole Farm Model-based assessments demonstrated that southern areas will face reduced profitability, whilst there will be a greater prevalence of arable farming in the northern parts of Europe. Interestingly all of the scenarios considered tended to increase the level of European production due to the fertilising effect of CO_2 and new areas becoming suitable for cropping.

The Whole Farm model was also applied to explore a range of agricultural futures for farming in England and Wales, through a comprehensive assessment that considered the environmental impacts of contrasting scenarios (e.g. “world-markets” vs “domestic self-sufficiency”). The results highlighted that although intensive farming under a “market driven” scenario resulted in higher environmental burdens within farmed areas, the increase in production intensity could potentially allow some land to be used for other purposes such as woodland establishment or extensive farming (Morris *et al.*, 2005).

Despite the clear potential for the model to be used in considering future scenarios, Annetts and Audsley (2002) highlight that the development of appropriate data to describe the environmental burdens associated with every possible crop, operation and machinery option has been a difficult process, and there is a need to explore the interactions between environmental effects and profitability in more detail.

3.8.4 Land Use Allocation Model (LUAM)

The LUAM is a GAMS[8] based linear programming model, which includes a spatial dimension by taking into account the historical patterns of production on different land classes. The objective function (i.e. the value to be maximised) is based on the net margin contribution of agricultural activities, which are adjusted by including a set of weights which alter the financial contributions of each enterprise on the basis of the historical persistence of the given enterprise in each land area/region. Therefore an enterprise which is particularly common in a particular land area will contribute more to the net margin than in a land area where it is rarer. The weights are taken to represent, without actually defining them, a range of factors that constrain the production of enterprises to certain areas. These can include factors such as levels of skills and expertise, tradition, soil type, capital endowments and complementarities with the production of other enterprises (Jones and Tranter, 2007).

The land base of the model, and the associated weights, are characterized using an adapted version of the Centre for Ecology and Hydrology's (CEH) Land Classification System (LCS), summarised in Table 3.7 below:

[8] General Algebraic Modelling System: www.gams.com

Table 3.7: Summary of the land classes used within the LUAM. Source: Jones and Tranter (2007):

No.	Name	Description
1	England, Central South downs	Gently rolling country with moderate relief. Rich farmland, both pastoral and arable
2	England, South-east downs	Long rounded slopes, particularly associated with chalk downs
3	East Anglia central plains	Almost flat plain with extensive arable farming
4	East Anglia Marginal plains	Intensively farmed arable: flat featureless plain
5	England, central plains	Undulating landscape with many hedges: mainly pastoral, often with heavy soil
6	England, South-west lowlands	Low emphasis relief with local variation. Predominantly pastoral agriculture
7	England/Wales coastal	Widely variable with estuaries, cliffs or steep slopes and associated lowland land, occasionally marsh or dunes
8	Midlands and North Wolds of England	Undulating land mainly with arable valleys and slopes
9	North East Wolds of England	Undulating land, predominantly pastoral and woodland
10	England, Midlands plains	Gently undulating land with intensive arable farming, except on heavy soils
11	England, North-West Midlands and North Lowlands	Intensively farmed: Level, featureless alluvial plains and coastal sites with mud flats: Intensive arable agriculture
12	Lowlands of Wales and Northern England	Intensively farmed flat land, inland from coasts
13	Wales central uplands	Gently undulating hills with moderate relief. Mainly pastoral agriculture
14	Northern England	Somewhat steeper slopes, and higher altitudes than 13 above. Less farmed land available, some forestry
15	East Lowlands of England	Intensively farmed pasture; gently rolling slopes with some upland influence.

The LUAM allocates land-use within each region according to the economic margin that can be achieved in relation to input costs (e.g. manufactured fertiliser and labour) and agricultural land-availability, with each activity contributing to the total net margin. The model therefore treats the entire agricultural area of England and Wales as if it were a single farm, with different conditions across the 'farm' (e.g. relating to soil type and pest and disease vulnerability) expressed through the regional weightings.

The LUAM model has been applied in a number of policy-focussed studies exploring the impacts of changing demand and product prices on agriculture in England and Wales. For example Arnoult *et al.* (2010) applied the LUAM to estimate the effects of a move towards healthier diets on agricultural land, finding that such a transition could improve the economic performance of agriculture in England and Wales, through an increase in higher economic-value crops (e.g. horticulture enterprises) and a contraction in enterprises with a lower-economic margin (e.g. beef and sheep).

In a separate study, an adapted version of the LUAM was used to assess the impacts of climate change on the geographical pattern of land-use in England and Wales, using price and commodity-demand data from a previous world-food trade study (Rosenzweig and Parry, 1994). The results illustrated that the projected economic change under a ‘no-climate change scenario’ could result in a reduction of the future area under agricultural production in England and Wales, as a result of increased crop yields and improved technology. The most serious climate change scenario resulted in a considerable change in cropping patterns, with reduced cereal cropping in some areas, as a result of higher temperatures and rainfall, and increased uptake of ‘new’ crops such as sunflower and grain maize. The results from this study also illustrated the importance of considering the economic impact of global markets when considering the national impacts of climate change (Hossell *et al.*, 1996).

3.8.5 Farm Modelling Information System - FARMIS

The Farm Modelling Information System (FARMIS) is a comparative-static process-analytical programming model based on Farm Accountancy Data Network (FADN[9]) data, with individual farms’ data being aggregated to farm groups to ensure confidentiality and increase the robustness of the modelling system (Küpker *et al.*, 2006b). The model was developed at the Johann Heinrich von Thünen-Institute (VTI –Federal Research Centre for Agriculture) in 1996 and first used for policy analysis in Germany in 1998 (Schader, 2009). The FARMIS system enables the assessment of different policy options at the regional and farm group level, through an optimisation framework (Küpker *et al.*, 2006b). Comparative-analytical and mathematical processing is applied to farm

[9] The FADN is an instrument for assessing incomes of agricultural holdings within the EU by farm type and the impact of Common Agricultural Policy measures. In England and Wales, FADN data is collected via the FBS.

groups, to optimise an objective function, subject to a set of resource and policy constraints, with the main objective function being to maximise farm income[10], subject to the (opportunity) costs for land and labour and interest on borrowed capital (Brockmeier and Urban, 2008; Küpker *et al.*, 2006b). Data extracted from the FADN are structured and organised within a Structured Query Language (SQL) database, which serves as the main source for the farm model.

The constraints set within the model cover a number of areas including feeding (energy and nutrient requirements, calibrated feed rations), intermediate use of young stock, fertiliser use (organic and mineral), labour (seasonally differentiated), crop rotations and institutional restraints (e.g. set aside, quotas) (Küpker *et al.*, 2006b). More recently the model has been expanded to include the trade of land, milk quotas and premium rights to help determine the supply and demand of production factors and determine equilibrium prices (Brockmeier and Urban, 2008).

A total of 27 crop and 15 livestock activities form the core of the model (Küpker *et al.*, 2006a). The model calibration is achieved through a Positive Mathematical Programming (PMP) approach, which applies country specific calibration factors extracted from an LP-based assessment of a 'baseline-year'. The PMP model is then used for future projections on factors such as land-use, production, and different income indicators (Brockmeier and Urban, 2008). The standard FARMIS procedure is summarised in the following steps (Schader, 2009):

1. Farm groups are assembled using FADN data
2. Input-output data are generated for each farm group
3. Detailed model assumptions are specified in order to address the research question being investigated
4. The model is calibrated for the base year by running it as a linear program with calibration constraints

[10] Farm income here refers to the Farm Net Value Added (FNVA). Costs of fixed factors are covered within this calculation, whether or not they are owned by the farmer (Küpker *et al.*, 2006).

5. Policy scenarios are calculated using a calibrated PMP model and scenario-specific assumptions

Policy simulation using the model takes place through the establishment of a ‘reference scenario’ for a target year in the future (i.e. usually assuming that the present agricultural policy and farm practices will continue). Alternative policy measures are then specified through additional activities or restrictions (e.g. changes to matrix coefficients). The outcome of this is then compared to the reference scenario (Schader, 2009).

Küpker *et al*. (2006a) used FARMIS to explore the effects of decoupling levels of production from support payments, within the 2003 CAP reform, finding that the decoupling of support payments in France and Germany would reduce acreages of major crops such as cereals, oilseeds and protein crops, and fodder maize, due to a loss of relative economic attractiveness. Küpker *et al*. (2006a) also found that energy crops would shift from being grown on set-aside to non-set-aside land, indicating that food crops would be replaced with biofuels on productive arable land. The authors also found that ‘pure’ set-aside (i.e. land without crops) would increase by 21% under the reforms. In addition the number of bulls and suckler cows was found to reduce in Germany, although numbers would increase in France as the reforms would encourage the maintenance of less-intensive systems in disadvantaged areas. The authors of this study also highlight the need for improving the FARMIS model with regard to balancing the sales/movements of young animals at an EU level or at least across several (neighbouring) countries, to allow the implications of trade-flows of young-stock to be incorporated within the modelling. The authors also point out that extensive activities need to be better- formulated within the model (Küpker *et al.*, 2006a).

3.8.6 Common Agricultural Policy Regionalised Impact Analysis (CAPRI)

The CAPRI model was established in 1999 with the objective of assessing the effect of Common Agricultural Policy instruments at a member-state and sub-national level. CAPRI was first applied to assess the ‘Agenda 2000’ reform package of the Common Agricultural Policy (Britz and Witzke, 2008). The model is economic in focus and consists of the following sections:

1. Specific databases, which extract information on cropping areas and outputs from well documented and official harmonised sources
2. An assessment methodology description
3. The software itself, used to complete assessments

The model is formed of two sections a ‘supply module’ and a ‘market module’. The supply side covers all variable costs and income, with low/high-yield variants. Approximately 50 inputs and outputs are included within the model; a high level of detail is provided on NPK balances, premiums paid under the CAP feeding activities and nutrient requirements of animals (Britz and Witzke, 2008). The database connected to the model is derived from various sources such as national statistics on slaughtering, herd size, crop production, land use, farm and market balances and foreign trade as well as regional statistics derived from the REGIO database[11] (Weiss and Leip, 2012). The model comprises about 50 crop and animal activities for approximately 280 regions at the Nomenclature of Territorial Units of Statistics Level 1 (NUTS 1). Each model within CAPRI maximises regional agricultural income at a given price and level of subsidy, subject to constraints on land, policy variables and feed and plant nutrient requirements in each region. The income is calculated as the sum of crop/livestock gross margins minus a quadratic function and subjected to a land availability constraint. Behavioural functions can also be implemented within the model, in order to capture the aggregated influence of economic factors that are not explicitly included in the original model (Jansson and Heckelei, 2011). The CAPRI model can therefore include an application of econometric assessment techniques. The CAPRI model is therefore both a database and a simulation model for the agricultural sector of the EU, and integrates economic, physical and environmental information in a consistent way (Weiss and Leip, 2012). In order to make a broader assessment of the effects of agricultural policies, single

[11] https://unstats.un.org/unsd/progwork/pwform.asp?TitleId=369

farm data are grouped into representative farm types. The sum of regional farms then represents the agricultural sector of a certain geographic area.

Within CAPRI, information on GHG fluxes for all emission sources from agriculture are calculated according to the IPCC guidelines (2006). Fluxes of nitrogen are estimated using a mass-flow approach developed for the MITERRA-EUROPE model (Weiss and Leip, 2012). The CO_2 emissions from energy use are based on Kränzlein (2009) whereas CO_2 fluxes and carbon sequestration rates are based on Soussana *et al.* (2007). The functional units vary depending on the process being measured; sometimes emissions are reported on a headage basis or per hectare of land, whereas for processes such as transport they are calculated on per product basis (e.g. CO_2 emissions from transport of feed/crops). Emission intensities are carried through to the end product, following pre-defined allocation rules. A team of researchers, based at the University of Bonn manage the model and are responsible for updates and distribution. There are currently no-fees for its use, but entry into the CAPRI network is controlled by the members. CAPRI has its own website, which provides information on the development of the tool to date and its potential use: http://www.capri-model.org/

The CAPRI model was used by a team of researchers at the Joint Research Centre in Italy to calculate total Life Cycle emissions from the EU livestock sector. The assessments found that 28-29% of CO_2 equivalents within the sector are from beef production, 28-30% from cow milk production and 25-27% from pork production, although large ranges were found in the emission intensities across the EU-27 (Weiss and Leip, 2012). The model was also used to determine farm, landscape and soil level nitrogen budgets/use efficiency and nitrogen surpluses for agricultural systems within individual countries in Europe, showing that countries that imported a high amount of feed were not able to effectively utilise the manure excreted by the animals, resulting in inefficient nitrogen use (Leip *et al.*, 2011).

3.8.7 Selection of models for this study

A number of the modelling approaches discussed in section 3.8, could be applied within a study exploring the impacts of a 100% conversion to organic farming, however the suitability of each will depend on the extent to which they meet the following success criteria, defined in the context of this study:

1. The approach or models chosen should be designed for or readily adaptable to UK conditions
2. The approach should allow for an assessment of the range of indicators to be used within this study (i.e. production, greenhouse gas emissions, fossil energy use)
3. The approach should allow for interactions between farming systems (e.g. sale or transport of livestock feed or manure)
4. The approach should be able to consider effects of change in practice both within the UK and overseas, to avoid the perception that a change in practice is effective, when in fact the change only causes displacement activity (e.g. land use change)
5. The approach should already be suited to or readily adaptable to organic modes of production
6. The models/modelling approach should be readily available (e.g. free to access/download and available within the UK)

Each of these success criteria are considered for each of the models in Table 3.8.

Table 3.8: Review of modelling approaches discussed in section 3.8

Model name	Suitability for UK conditions	Capture of full range of indicators to be used in this study	Allows for interaction between farms (e.g. trade effects)	Consideration of global impacts	Suitability for organic systems	Availability
Agri LCA models	Designed for UK conditions	Covers GHG and energy use (i.e. environmental impacts)	Yes	Yes, takes an LCA approach	Has already been used to assess environmental performance of organic farms in a Defra project	Freely available for download at: www.agrilca.org
Silsoe Whole Farm Model	Designed for UK conditions	Crop yields and N use and leaching are included	No but LP approach could be adapted	No	Not included in the current model	Available from Cranfield University
LUAM	Designed for UK conditions	Not directly – reports land use and production	Yes	No	Not included in the current model but could be adapted	Available from Reading University
FARMIS	Europe-wide in focus but could be adapted to UK	Focuses on economic impacts, energy use is included as a separate category	Yes - trade of land, milk quotas and premium rights	Not within original model but recent work has included this (i.e. Schader (2009)	Has already been used to assess organic systems in Switzerland (i.e. Schader, 2009)	Model is freely available from the von Thünen Institute (VTI) in Germany

Model name	Suitability for UK conditions	Capture of full range of indicators to be used in this study	Allows for interaction between farms (e.g. trade effects)	Consideration of global impacts	Suitability for organic systems	Availability
IMPACT	World-wide focus	No, includes crop yields but not energy use, greenhouse gases or Soil Carbon	Yes, captures net trade at national and international level	Yes, global impact with focus on food security	Not included in the current model	Cost for access from IFPRI (based in Washington DC)
CAPRI	European in focus	GHG fluxes including CO_2 from fossil fuel	Yes, captures net trade at national and international level	No European focus	Would need to be adapted for organic production systems	Request to access the tool can be made through the University of Bonn

Based on the comparison of the modelling approaches (Table 3.8) a linear-programming (LP) method was selected to determine the production impacts of a 100% conversion to organic farming. In addition to being a well-developed method for the multi-factorial assessment of UK agricultural systems (see sections 3.8.3 and 3.8.4) the flexibility inherent to LP methods allows for a range of scenarios to be assessed in a systematic and efficient manner and for the identification of trade-offs between management choices (Annetts and Audsley, 2002). An LP model was combined with the Agri-LCA to determine domestic and overseas environmental impacts of the production scenario.

The approach built on the development of the LUAM, which was constructed in GAMS by Philip Jones and colleagues at Reading University (see: Arnoult *et al.*, 2010). GAMS-based coding offers several advantages over other common programming languages. In particular by allowing for simple model classification through the use of mathematical symbols and algebraic relationships it is possible to define models in unambiguous terms that can be readily understood and adjusted. In addition the language includes in-built features for error-detection and has direct compatibility with Excel through a GAMS Data Exchange (GDX) facility. All identifiers used within the GAMS code must also be declared and described with associated text before being referenced in the model, which can be of great assistance when returning to models after a period of absence in addition to helping with understanding of the modelled processes and interactions. The large library of existing models and an online GAMS community (http://www.gamsworld.org) also allows for sharing of ideas and approaches between those facing similar challenges (Bussieck and Meeraus, 2004). The Centre for Agricultural Strategy at Reading were also willing to engage and assist with an LP / GAMS modelling approach which was a key influencing factor in the selection of this method.

CHAPTER 4. ENERGY EFFICIENCY IN ORGANIC FARMING

Article title	The energy efficiency of organic agriculture: A review
Journal and publication status	*Renewable Agriculture and Food Systems.* Published
Co – authors	Adrian Williams (Cranfield University), Bruce Pearce (The Organic Research Centre)
Co-author contributions	Guidance on planning of content including search strategy, provision of literature
Research methods applied	Structured literature review
	Systematic data collection and analysis

4.1 Abstract

Growing populations and a constrained fossil-manufactured energy supply present a major challenge for society and there is a real need to develop forms of agriculture that are less dependent on finite energy sources. It has been suggested that organic agriculture can provide a more energy efficient approach due to its focus on sustainable production methods. This review has investigated the extent to which this is true for a range of farming systems. Data from about 50 studies were reviewed with results suggesting that organic farming performs better than conventional for nearly all crop types when energy use is expressed on a unit of area basis. Results are more variable per unit of product due to the lower yield for most organic crops. For livestock, ruminant production systems tend to be more energy efficient under organic management due to the production of forage in grass–clover leys. Conversely, organic poultry tend to perform worse in terms of energy use as a result of higher feed conversion ratios and mortality rates compared to conventional fully housed or free-range systems. With regard to energy sources, there is some evidence that organic farms use more renewable energy and have less of an impact on natural ecosystems. Human energy requirements on organic farms are also higher as a result of greater system diversity and manual weed control. Overall this review has found that most organic farming systems are more energy efficient than their conventional counterparts, although there are some notable exceptions.

4.2 Introduction

Non-renewable (mainly fossil) energy inputs have played an important role in increasing the productivity of our food systems and sustaining the exponential rise in the world's population witnessed over the last century (Smil, 2000). At the same time, the dramatic rise in production levels required to support increased populations has created a dependence on mined sources of so-called "stored-solar energy" (Hall, 1984) within the developed world. This, in turn, has led to agricultural systems that are more exposed to fluctuations in the prices of fossil fuels, whether caused by political instability or increasing demand. A range of environmental catastrophes caused by the pursuit of ever-more scarce sources of fossil energy have also caught the media and public's attention in recent years. These include the Deepwater Horizon oil rig disaster in 2010 and the Exxon natural gas project disaster in Papua New Guinea in 2012. Such events have served to underline the risks associated with our reliance on these energy sources (Trevors and Saier, 2010). With this growing awareness, our vulnerability and the continuation of 'agri business as usual' has been questioned (McIntyre *et al.*, 2008).

In this context, organic agriculture has evolved as a farming system that focuses on the preservation and recycling of resources, with the aim of creating more sustainable production systems (IFOAM, 2002; Kukreja and Meredith, 2011; Lampkin, 2002; Lampkin *et al.*, 2011). This has been encouraged through the development of an underlying set of internationally accepted principles, and legally binding standards in some jurisdictions, that define organic agriculture (Darnhofer *et al.*, 2010; European Commission, 2008; Soil Association, 2008; United States Department of Agriculture, 1990). With the focus on reducing inputs within the organic sector, it should follow that the adoption of organic production methods will result in farming systems that are less dependent on fossil fuel inputs. Recent reviews by Lynch *et al.* (2011), Gomiero (2008) and Lampkin (2007) report that organic agriculture consistently has lower energy use and greenhouse gas emissions when results are expressed on a per hectare basis. Results were more variable when presented per kilogram of product, and conventional production was found to have the highest levels of net energy production. The above studies also found that the variety in energy assessment methods make direct comparisons between studies difficult. The magnitude of difference between organic and conventional production varied greatly depending on whether 'conventional' production within a given region is of an intensive or extensive nature (Lynch *et al.*, 2011).

The aim of this review is to build on the work of Gomiero (2008), Lynch (2011) and Lampkin (2007) by assessing the results from studies comparing the energy use and energy efficiency of organic and conventional farming systems. Unlike previous work, the review presented here provides an overview of energy use according to the type of input (e.g. fuel for machinery, embodied energy in feed and fertiliser). A more complete overview of studies that have considered the embodied energy associated with inputs and ecosystem services is also presented (i.e. results from emergy studies). In addition, the results from more recent published work have been included here. This review also explores the extent to which the results from these studies vary according to the scope of the assessment, the unit of measurement and the farm or production system.

4.3 Methods

A structured literature review of organic/conventional energy use studies was carried out by the lead author in 2012 using a range of web based search engines (ISI Web of Knowledge, Scopus, Google Scholar, BIOSIS Previews, SCIRUS , ScienceDirect, Organic Eprints). The following or similar terms were used in a combination with the Boolean operators AND, OR:

- energy, emergy, fossil fuel
- organic, biodynamic, agro-ecological
- life cycle assessment, LCA, emergy, thermodynamic
- comparison, compare

Only studies based on pairwise comparisons were selected for inclusion and publications had to contain energy use data on both organic and conventional agriculture. Non-certified production systems were also included, for example where experimental farms were using organic methods. In these cases a judgement was made as to whether the farming practices on the experimental farm being assessed adhered to the IFOAM (International Federation of Organic Agriculture) principles. Countries in the developing world were excluded and the review focussed on modern agricultural systems (e.g. excluding the use of draught animals for cultivation). Studies compared were drawn from Europe, North America, Canada, Australia and New Zealand. Grey literature was included within the search, including PhD theses**,** Government and NGO reports and research project reports. A total of 48 studies were identified, as shown in Table 4.1. Although the approach taken within this study did not follow published guidelines for the completion of systematic reviews, the search strategy was comprehensive and the results provide an effective and thorough assessment of relative performance within a range of organic systems.

Energy use data were extracted from each paper and placed into Excel to allow for a comparison between organic and non-organic production systems and individual products. Production systems were grouped into the following categories for the purpose of this comparison:

- Cropping
- Dairy
- Beef and sheep
- Pig and poultry
- Vegetable and fruit
- Other

Drawing on this literature, comparisons were made in relation to the amount of energy required per unit of product (e.g. kilograms or litres) in addition to the amount used per unit of land (e.g. hectares or acres) within each farm-type. This approach follows the suggestion of Van der Werf *et al*. (2007a) who propose that the unit of area comparison reflects a farming system's function as a producer of non-market goods (e.g. biodiversity) whereas the unit of product comparison reflects agriculture's function as a producer of market goods (e.g. food and fuel). Comparisons of environmental performance based solely on the amount of product can also present an issue when dealing with foodstuffs that vary greatly in nutritional and water content (e.g. milk and meat, De Vries and

De Boer, 2010). Furthermore, Cherubini and Strømman (2011) highlight that displaying results per unit of agricultural land can provide a useful indicator of land-use efficiency. The same study highlights the need to identify the limiting factor of the system being assessed and that this should be used as the reference indicator of the assessment. With competition for agricultural land proposed to be one of the main drivers affecting food and farming in the future (Foresight, 2011) assessing energy use per unit of land can be a useful tool to compare the energy efficiency of agricultural systems.

4.3.1 Types of study considered

Most of the studies considered within this review have taken what Jones (1989) describes as a ‘mechanistic’ or ‘process analysis’ approach, i.e. assessing the fossil energy use associated with the various production stages of an agricultural product. This includes the assessment of energy associated with production processes on case study farms (Alföldi *et al.*, 1999; Cobb *et al.*, 1999) or through the application of Life Cycle Assessment (LCA). This is a method used to calculate the burdens associated with one unit of a food commodity, e.g. 1 kilogram of wheat, area of land or Livestock Unit (LU) defined as the ‘functional unit’ (British Standards Institute, 1997). Within the LCA approach, inputs to the system are usually traced beyond the farm gate to the primary resource. For example, this can include the coal or uranium used to generate electricity or the energy required to produce steel, plastic and other materials required for the manufacture of tractors (Williams *et al.*, 2006). LCA has the distinct advantage of being able to determine efficiency within supply chains in a manner that can be easily understood (Wegener Sleeswijk *et al.*, 1996). In addition, the broad principles for the application of LCA have been standardised, e.g. through the the International Organization for Standardisation 14044 standard (British Standards Institute, 2006). This has helped to make LCA the most widely used method for the assessment of energy use within supply chains in the agriculture sector (Pelletier *et al.*, 2011). It is important to note however that these standards are not prescriptive about boundary conditions, the functional unit or the purpose of the study, which can make comparisons between studies difficult.

Other studies considered here have followed a ‘thermodynamic approach’ (Jones, 1989) through the adoption of emergy accounting (Odum, 1996). Emergy has developed as an alternative to the ‘traditional‘ fossil energy focussed approach of energy accounting. It takes an eco-centric approach that accounts for the contribution of natural services (e.g. rain, pollination, soil formation) in delivering agricultural products (Bakshi, 2002). In a similar manner to LCA, the emergy approach measures the energy previously used in the creation of a product. However it also accounts for the amount of available energy that sits within the assessed product or system. The units of energy are expressed in a common unit (i.e. ‘solar energy’ or ‘emjoules’, Odum, 1996). The emergy approach also takes into account natural/ecological inputs and human activities. It calculates natural inputs, based on the distribution of solar energy in the biosphere and the energy output potential of the various processes (e.g. rainfall, total wind energy, total wave energy, Brown and Herendeen, 1996). Human labour and services can also be accounted for, both in terms of the energy used to support human life and the energy associated with the accumulation of information (Odum, 1996). In this sense, emergy allows for an assessment of ‘energy quality’ through considering the importance of inputs and outputs in a web of relationships (Pizzigallo *et al.*, 2008). A limited number of studies have used the emergy approach to assess the efficiency of organic and conventional agriculture. The results from these studies will be described in a separate section below.

A number of studies within this review have also taken the nominally dimensionless ‘energy ratio’ approach to determine the efficiency of production systems (i.e. dividing the energy output in food sold by the energy input of fossil fuels). This approach is nominally dimensionless in that the gross energy of fuels is compared with the metabolisable energy of foods or feeds. 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.

A limitation of the study is that there is insufficient data to perform a statistical analysis. The wide variation in the scale of the studies and the methods used prevents this. In addition, the wide geographical variation in the studies, and the resultant wide range of soil types and climates, makes it difficult to draw definitive conclusions that will apply to each country or region (Table 4.1 shows the list of studies, their location and the energy assessment method used).

4.4 Results from the literature survey: studies selected for inclusion

Most of the comparative studies listed in Table 4.1 were based on production systems operating in Europe although some studies from the USA, Canada and Australasia were included.

Table 4.1: list of studies selected for inclusion within the review

Author of study	Country	Production system/farm types	Method
Alonso and Guzmán (2010)	Spain	Vegetables, arable crops, fruit	Input/output assessment using farm data
Bailey et al. (2003)	UK	Arable	Data collected from 5 years of field trials
Basset Mens and van der Werf.2005	France	Pigs	Modelling of farm systems using published and expert data using LCA
Mäder et al. (2002)	Switzerland	Cropping	Experimental farms
Bos et al. (2007)	Netherlands	Arable, dairy, vegetables, mixed	Direct/indirect energy use modelling (did not follow ISO LCA standard)
Cederberg and Mattsson (2000)	Sweden	Dairy	LCA using measured farm data and published data
Clements et al. (1995)	Canada	Arable	Experimental farm data and farm survey
Cormack and Metcalfe. (2000)	UK	Arable, dairy, vegetables, beef and sheep, mixed	Modelling based on book values
Dalgaard et al. (2001)	Denmark	Arable, dairy and pig production	Direct/Indirect Energy Modelling
Deike et al. (2008)	Germany	Arable	Long term field experiment
Flessa (2002)	Germany	Beef and arable	Experimental farm data
Geier et al. (2001)	Germany	Apple production in Hamburg (organic intensive, organic extensive and integrated)	LCA using farm data and published data
Grönroos et al. (2006)	Finland	Dairy	LCA using statistics and expert opinions
Gundogmus (2006)	Turkey	Raisin	Structured interviews & direct/indirect energy model
Guzmán and Alonso (2008)	Spain	Olive oil production	Calculated energy balances using data collected through farmer interviews
Haas et al. (2001)	Germany	Dairy	LCA using published agricultural planning data
Helander and Delin (2004)	Sweden	Arable	Results from research farm-based comparison
Hoeppner et al. 2005	Canada	Arable	Crop rotation experiment
Kaltsas et al. (2007)	Greece	Olive oil production	LCA using data collected through interviews
Karlen et al. (1995)	USA	Arable	Farm level comparison
Kavargiris et al. (2009)	Greece	Grapes	Energy analysis using data collected through farmer interviews
Klimeková and Lehocká (2007)	Slovakia	Spring Barley	Field experiment data

Author of study	Country	Production system/farm types	Method
Kusterman et al. (2008)	Germany	Arable	REPRO model and data collected from experimental farm
Leinonen et al. (2012a)	United Kingdom	Poultry - meat, standard	LCA, structural model of industry
Leinonen et al. (2012b)	United Kingdom	Poultry - eggs, caged	LCA, structural model of industry
Mäder et al. (2002)	Switzerland	Farm comparison - Conventional FYM / Biodynamic	Data collected from experimental farms
Meisterling et al. (2009)	USA	Wheat	LCA modelling study
Nemecek (2005)	Switzerland	Arable	Swiss Agricultural Life Cycle Assessment method (SALCA)
Nguyen and Haynes (1995)	New Zealand	Arable and livestock	Farm comparison
Pelletier et al. (2008)	Canada	Wheat	LCA Scenario modelling
Peters et al. (2010)	Western Australia	Beef and sheep	LCA using public/published data
Pimental et al. (1983)	USA	Arable, Apples	Modelling based on published data
Pimental et al. (2005)	USA	Arable	Recorded energy use from experimental farm at The Rodale Institute
Refsgaard et al. (1998)	Denmark - non irrigated sand	Arable, Dairy, forage	System modelling using farm data from Government survey
Reganold et al. (2001)	USA	Apples	Farm comparison
Schader (2009)	Switzerland	Arable, beef, sheep, dairy, vegetables, poultry, pigs, mixed	LCA using farm and public/published data
Thomassen et al. (2008)	Netherlands	Dairy	LCA using public/published data
Van der Werf et al. 2007	Brittany, France	Pig production	Modelling of farm systems using published and expert data using FarmSmart tool
Venkat (2012)	USA	Arable, vegetables, fruit, nuts	LCA modelling using production data
Williams et al. (2006)	United Kingdom	Arable, beef, sheep, dairy, vegetables, poultry, pigs, mixed	LCA using public/published data
Williams et al. (2010)	UK	Arable	LCA using public/published data
Wood et al. (2006)	Australia	Sheep, arable, vegetables, fruit	Hybrid LCA incorporating a farm survey
Castellini et al. (2006)	Italy	Poultry - meat	Emergy
Pizzigallo et al. (2008)	Italy	Wine production (including processing post farm-gate)	Emergy and LCA
La Rosa et al. (2008)	Italy	Red orange production	Emergy
Coppola et al. (2009)	Denmark	Wheat	Emergy
Ghaley and Porter (2013)	Denmark	Wheat and combined food and energy system comparison	Emergy

4.5 Results from the literature survey: on farm energy use

The efficient use of fossil fuel energy on farm is of increasing concern for farmers and stakeholders within the supply chain, in light of fluctuating input prices (Cassman and Liska, 2007; Woods *et al.*, 2010) and the effects of climate change and pollution (Smil, 2000). A number the process-oriented and LCA-based studies listed in Table 4.1 compared on farm resource efficiencies within a range of organic and conventional crop and livestock systems. In addition, a number of studies have assessed human energy, using empirical methods or system modelling; the results from studies in both of these areas are outlined below.

4.5.1 On farm fuel use

A common criticism of organic agriculture is that a reliance on mechanical tillage (e.g. for weed control) results in lower energy efficiency overall (Hoeppner *et al.*, 2006). A process oriented modelling study carried out by ADAS (Cormack and Metcalfe, 2000) supported this criticism, finding higher machinery energy use within organic systems (i.e. energy associated with the manufacture, distribution and repairs to mechanical equipment). This increase was, however, offset by higher indirect energy use under conventional management. Most of the additional fuel use within the ADAS study was associated with weed control. Organic carrot production compared particularly poorly due to the energy intensive process of flame weeding. Organic wheat production was also associated with higher machinery energy, a potentially significant finding in view of the dominance of wheat in the European arable sector and the importance of this crop as a staple of Western diets. Venkat (2012) also found higher on farm energy use on organic farms for certain vegetable crops such as broccoli (see Figure 4.1) within an LCA comparison, suggesting this is due to systematically higher levels of mechanical weeding. Unlike the ADAS study, Venkat (2012) found that this difference was enough to offset the impact of fertiliser manufacture in the conventional system. Greater use of tractor diesel per litre of milk produced was also reported for an organic farm in an LCA of two large dairy units in Sweden (Cederberg and Mattsson, 2000). Higher fuel use per 1000kg of milk on the organic farm was a result of the larger area of fodder production and lower yields. Jorgensen *et al.* (2005) also found that levels of on farm energy use were 28% higher for organic crop production in Denmark. This was a result of higher fuel consumption for weed control in addition to the energy intensive practice of manure spreading (compared to spreading fertiliser). In common with the ADAS study, the authors found that the higher on farm energy use was offset by the energy requirements for the manufacture of inputs in the conventional system.

The need for mouldboard ploughing in organic systems, for the removal of crop residues and control of weeds, can also contribute to greater on farm energy use in comparison to reduced tillage with herbicides, as identified in a comparison of organic, integrated and conventional farming systems (Leake, 2000). A study by Michigan State University also found a lower fuel use for a corn, soybean, wheat rotation under conventional no till, compared to the same rotation under low input and organic conditions, although the savings were offset by the energy associated with fertiliser and lime inputs (Robertson *et al.*, 2000). Zentner *et al.* (2004) also found that gains in on farm fuel use from reduced tillage were offset by the embodied energy associated with inputs of pesticide and fertiliser within an energy analysis of nine cropping systems in Canada. Despite this Snyder and Spaner (2010) note that high input costs are supporting a shift toward reduced input systems, where reduced tillage is applied, and it has been suggested that such tightly controlled conventional systems may rival organically managed farms with regard to energy efficiency, even when the costs of inputs are taken into account (Clements *et al.*, 1995). Reduced tillage is no longer exclusive to conventional farms however. Recent studies show that this technique can be applied successfully under organic conditions for cereal crops (Berner *et al.*, 2008; Crowley *et al.*, 2012) with significant energy savings as a result (Crowley *et al.*, 2012). Lockeretz *et al.* (1981) also found that organic farmers in the ‘corn belt’ of the United States were more likely to use chisel ploughing methods, as opposed to the mouldboard plough. This was to help conserve organic matter and water, instead of exposing the soil to wind erosion, a common problem in the area studied. It is also important to consider that reduced tillage is not always possible for farmers. The possibility for implementation will depend greatly on soil type, topography and the available power of the machinery (Bailey *et al.*, 2003). Increased herbicide leaching and greater populations of certain perennial weeds and grasses has also been reported in some reduced tillage systems (Locke *et al.*, 2002; Tuesca *et al.*, 2001) which could result in increased requirements for cultivation and fuel use to remove pernicious weeds. Reduced yields within no tillage systems have also been observed on some soil and climate conditions (Lal *et al.*, 2007) reducing the overall energy efficiency per unit of product.

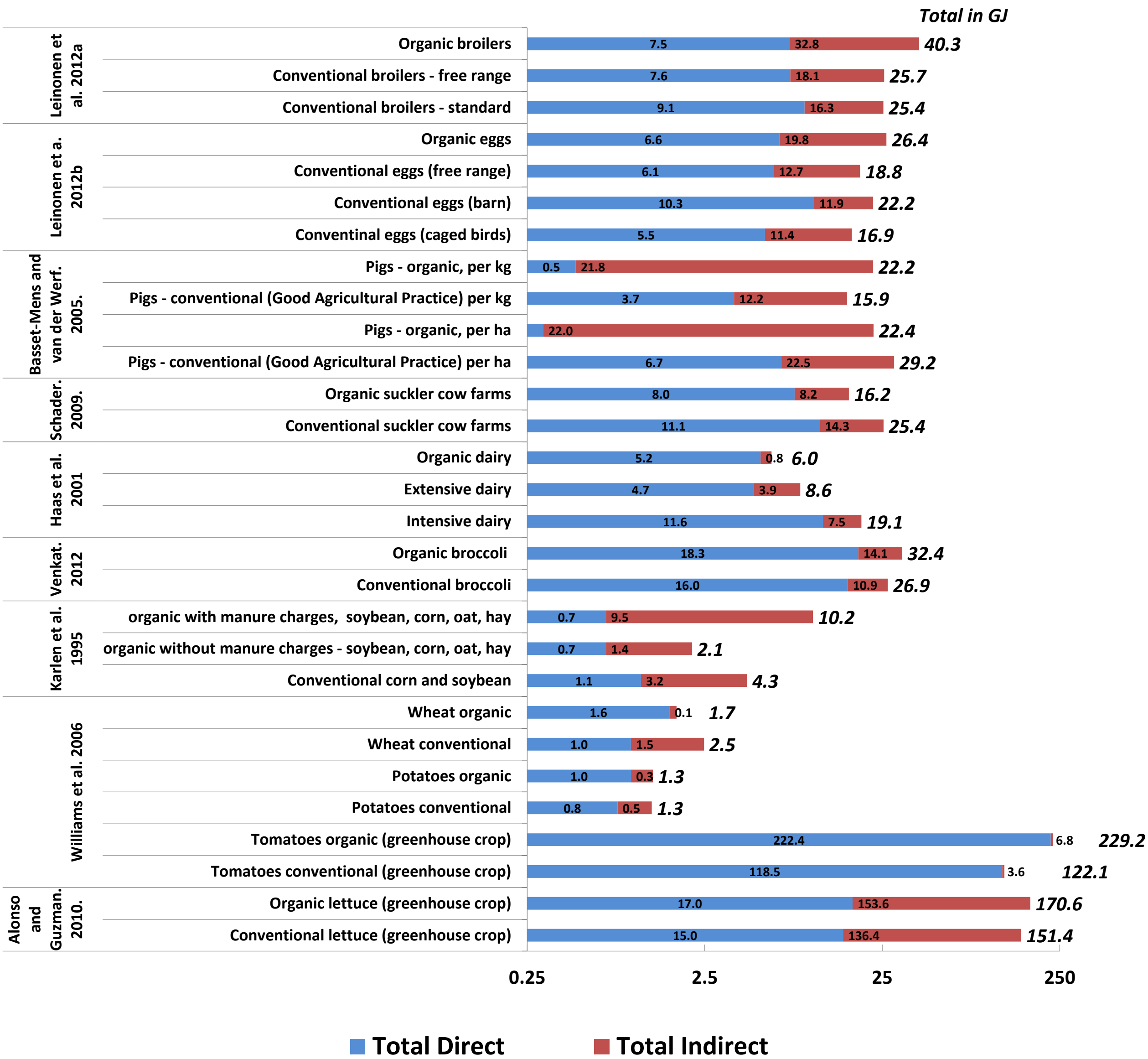

Figure 4.1: Distribution of 'direct' (i.e. on farm) and 'indirect' (i.e. off farm) energy use from nine studies comparing organic and conventional production. Due to variation in the scale for the products reported a log scale has been used on the x-axis. Most studies took a 'cradle-to-gate' approach (i.e. considering energy use associated with production but not retail consumption and disposal) for more details on boundaries and functional unit of each study, see Table 4.2.

In contrast to many of the above studies, some authors have found similar or even lower levels of on-farm diesel use for organic production. For example, Refsgaard *et al*. (1998) found little difference between the amounts of diesel required for the production of conventional and organic crops. However, the organic systems within the process models used in this study tended to require more fuel for handling and spreading of manure. A farm system monitoring project in Switzerland also found very similar levels of diesel in a long-term comparison of an organic and conventional farm, although the conventional systems used as a comparator within this study was of a relatively low intensity (Alföldi *et al.*, 1999).

4.5.2 Labour

With regard to human energy (or labour), organic systems have been associated with higher numbers of staff on the farm due to increased livestock, reduced machinery use and diversity in farm enterprises (Cobb *et al.*, 1999; Lobley *et al.*, 2005; Ziesmer, 2007). El-Hage Scialabba and Hattam (2002) also report that a higher share of labour intensive crops (e.g. vegetables) and on farm marketing and processing may lead to increased labour requirements on European organic farms. Within a modelling study of four organic and conventional crops, Pimental *et al*. (1983) also found lower labour productivity for organically produced crops (i.e. kg output per hour of labour input). This was due to a need for increased cultivations, in addition to greater losses from pests and disease and high cosmetic standards which prevent sale of certain crops, in particular organic apples.

Nguyen and Haynes (1995) also compared the labour productivity of three pairs of mixed cropping farms in the Canterbury region of New Zealand, with labour requirements calculated in hours per hectare for the entire rotation and the cropping part (i.e. peas, barley and wheat) separately. The labour productivity was also measured as a ratio of harvested grain to the number of hours per hectare. Although labour inputs per hectare for most grain crops were higher on the organic and biodynamic sites, the total labour use was lower as a result of the 3 to 4 year fertility building period. This balanced out the higher requirement for the cropping phase. Despite this, the grain crops grown within the biodynamic and organic systems had a lower labour productivity (0.4-1.1 tonnes/hour) compared to the conventional (1.3-1.6 t hr^{-1}), as a result of higher labour inputs and lower yields. The additional labour requirement within the organic systems was partly due to the additional field and manual operations plus the additional labour requirement for the manufacture of cow horn manure (a homeopathic preparation for improving soil health) within the biodynamic system. Karlen *et al*. (1995) took a similar approach in calculating the number of fieldwork hours required for crop production and harvest in a comparison of four 40 acre fields in the 'Corn belt' of the United States. Within the 'alternative' system, labour requirements were substantially increased (between 178% and 183% of the conventional). This was primarily as a result of the additional time required for spreading manure, weed control and through the incorporation of a hay crop within the rotation, which required multiple harvests.

An attempt was also made to compare the labour requirements of organic and conventional farms by comparing calendars of work for a conventional and organic farmer in addition to measuring heart rates and constructing an energy budget based on their food intake (Loake, 2001). The relatively high energy and effort expenditure on the organic farm led the author of this study to suggest that "the annual activity of organic farming is characterised by physical stress and fatigue". Unfortunately the study was flawed in that it compared an organic farmer using hand tools with a conventional livestock and arable farmer who spends most of the heart rate assessment period driving a tractor. The farms were therefore not comparable, and as the author notes, the organic farmer cannot be considered representative of the sector. Having said this, the study does contribute to addressing the methodological difficulties associated with comparing mechanised systems and manual operations.

4.6 Indirect, off farm energy use

Indirect energy use (i.e. energy use associated with the production and transport of inputs) typically exceeds on farm energy use within modern farming systems in developed countries, with fertiliser and imported feeds for livestock comprising the two major sources of energy inputs used for agricultural products (Pelletier *et al.*, 2011). The importance given to on farm or local resources within the IFOAM organic principles (Darnhofer *et al.*, 2010) suggests that organic farms could be less reliant on external inputs of fertility and animal feed, and a number of studies have explored the extent to which this applies in practice.

4.6.1 Fertiliser inputs

The energy intensive manufacture of nitrogen (N) based fertilisers represents the most energy expensive input for modern farming, accounting for about half of agriculture's energy use (Foresight, 2011) and approximately 1.1% of energy use globally (Dawson and Hilton, 2011). Instead of relying on manufactured fertilisers, organic farms source the bulk of their nitrogen through biological fixation by temporary, legume-based leys. The use of leys can also further the production of SOM (Leifeld and Fuhrer, 2010) in addition to providing an energy source for the soil biota, which enables humus production through transformation of organic material. In this, sense the organic system aims to develop soil health over the long-term, rather than providing a short term nutrient supply through application of soluble plant nutrients (Watson *et al.*, 2002). Refsgaard *et al.* (1998) state that in this context "one might think of organic farming as a systematic replacement of fossil fuel N fertilizer production with solar driven nitrogen fixation in legumes", with fossil fuels being used to help this process. This was illustrated by Gomiero (2008) who found that the main reason for increased energy efficiency under organic management was the lack of synthetic inputs, in particular fertilisers and pesticides.

Despite the reliance on biologically fixed nitrogen within organic agriculture, organic farmers still make use of mineral sources for other nutrients, in particular rock phosphate (P) which is mined from natural stores. Trewavas (2004) argues that when this aspect is taken into account, the energy efficiency of organic farming is lowered considerably, when compared to integrated no till systems. Low solubility of rock phosphate may also make it less effective than manufactured P fertiliser (superphosphate) particularly in low rainfall areas (Bolland *et al.*, 1988). Co-application of rock phosphate with elemental sulphur or manure could, however, help to enhance availability (Agyin-Birikorang *et al.*, 2007; Evans *et al.*, 2006). Use of rock phosphate may also help to maintain a stable supply of readily available P over time, compared to use of water soluble phosphate fertiliser (Randhawa *et al.*, 2006). Pelletier *et al*. (2008) found in their LCA of organic and conventional wheat and soy production in Canada, that the cumulative energy impacts of producing phosphate fertiliser were on average four times higher than those associated with producing rock phosphate used in organic agriculture. Sourcing fertility from outside of the farming system also applies to farms producing large quantities of crops, which depend on external sources of compost and manure. Alonso and Guzman (2010) for example found higher energy use for organic crops grown in Spain, as a result of the energy associated with production of large quantities of compost. Karlen *et al.* (1995) found that without charging for the energy associated with the manure nutrients (i.e. assuming that the manure is a 'cost' incurred by the livestock enterprise) an 'alternative' system required about half of the energy of the conventional, however if the energy costs for the nutrients were included, the alternative system used twice as much energy as the conventional (see Figure 4.1). Duesing (1995) in Rigby and Cáceres (2001) also refer to North Californian organic farmers using manure from South Californian dairy farms, which in turn used imported feed grain from the Midwest. Rigby and Cáceres (2001) note that such practices have serious implications in terms of energy use and that the methods used do not necessarily sit well with some people's perceptions of organic production, or the organic principles.

Despite evidence that some organic farmers are importing fertility and are therefore 'robbing Peter to pay Paul', Alonso and Guzman (2010) point out that inputs of manure and compost help to promote the long-term health of the system, and cannot be compared in the same way to non-renewable energy sources. They also highlight that organic farmers are able to reduce levels of compost application as soil humus levels develop. Moreover when a comparison was made of non-renewable energy use (i.e. fossil fuels) within this study, the energy use was significantly lower within all of the organic production systems. El-Hage Scialabba and Müller-Lindenlauf (2010) also highlight that the pollution and soil degradation problems associated with landless livestock production systems can be reduced through the co-operative use of farmyard manure between crop and livestock operations on organic farms. With landless livestock production systems currently supplying over 50% of pig and poultry meat worldwide (Steinfeld *et al.*, 2006) the relative advantages of a more integrated approach to production are an important consideration. Reviews comparing nutrient budgets on organic and conventional farms have also found that nutrient surpluses and nitrogen leaching are generally smaller for organic farms. This suggests a more efficient use and recycling of nutrients between enterprises (Shepherd *et al.*, 2003; Tuomisto *et al.*, 2012b).

4.6.2 Livestock feed

As mentioned above, organic farms try to maintain a closed production system as far as possible with regard to all inputs, not only those relating to soil fertility. Assessments of energy use within beef and dairy production by Schader (2009) and Haas *et al.* (2001) found that this approach manifests through a reliance on home grown sources of feed for livestock (see lower energy inputs associated with imported feed within these studies in Table 4.2). Lower energy use associated with concentrate feed has also been reported in comparisons of organic and conventional dairy production in Sweden, Denmark and the Netherlands (Cederberg and Mattsson, 2000; Jørgensen *et al.*, 2005; Thomassen *et al.*, 2008). Within an assessment of the environmental impacts of a 1996 'baseline' and a number of 100% organic conversion scenarios in Denmark, Dalgaard *et al*. (2001) also found that domestically produced, organic grass/clover was energetically cheaper than conventional forage, due to a lack of fertiliser application. The increased efficiency contributed to lower energy use overall per LU.

For poultry most organic production systems have longer production cycles. This can have a positive effect in terms of animal welfare (e.g. lower prevalence of limb disorders, through use of slow growing breeds, (Castellini *et al.*, 2008) but also results in lower energy efficiency through higher levels of feed use per unit of product (e.g. Leinonen *et al.* (2012a) see Figure 4.1). In addition, mortality rates of caged poultry systems have been shown to be lower than organic or free range systems (Leinonen *et al.*, 2012a, b). For pig meat production, recent studies have shown that organic systems tend to import less feed, which contributes to lower energy use and greater efficiency per unit of land, but also lower levels of output and a possible increased energy use per kilo of product, depending on the assessment method used (Basset-Mens and van der Werf, 2005; van der Werf *et al.*, 2007b). Williams *et al.* (2006) also reported a considerable increase in the area of land used for the production of pig feed within organic systems, in an LCA study of UK production. This led to a reduced energy output per hectare, compared to conventional production.

Table 4.2: Distribution of 'direct' (i.e. on farm) and 'indirect' (i.e. off farm) energy use from nine studies comparing organic and conventional production.

Author(s)	Product	Type of study	Unit	System boundary	Fuel and electricity	Purchased feed (indirect)	Fertiliser, compost, pesticides (indirect)	Machinery and buildings (indirect)	Other	Total
Leinonen et al. 2012a	Chickens - broilers - organic	LCA	GJ/tonne	Cradle to gate. Manure treated as energy credit due to fertiliser production offset	7.5	32.8	-0.5	0.5	0.0	40.3
	Chickens - broilers - conventional, free-range	LCA	GJ/tonne		7.6	18.2	-0.4	0.3	0.0	25.7
	Chickens - broilers - conventional, standard	LCA	GJ/tonne		9.1	16.4	-0.4	0.2	0.0	25.4
Leinonen et al. 2012b	Chickens - layers - organic	LCA	GJ/tonne	Cradle to gate. Manure treated as energy credit due to fertiliser production offset	6.6	19.9	-0.4	0.3	0.0	26.4
	Chickens - layers - conventional, free range	LCA	GJ/tonne		6.1	12.9	-0.5	0.3	0.0	18.8
	Chickens - layers - conventional, barn	LCA	GJ/tonne		10.3	12.1	-0.4	0.2	0.0	22.2
	Chickens - layers - conventional, caged	LCA	GJ/tonne		5.5	11.6	-0.4	0.3	0.0	16.9
Basset-Mens and van der Werf. 2005	Pigs - organic	LCA	GJ/tonne of pig	Cradle to gate	0.0	0.0	0.0	0.5	21.8	22.2
	Pigs - conventional	LCA	GJ/tonne of pig		0.0	0.0	0.0	3.7	12.2	15.9
	Pigs - organic	LCA	GJ/ha		0.0	0.0	0.0	0.3	22.0	22.4
	Pigs - conventional	LCA	GJ/ha		0.0	0.0	0.0	6.7	22.5	29.2

Author(s)	Product	Type of study	Unit	System boundary	Fuel and electricity	Purchased feed (indirect)	Fertiliser, compost, pesticides (indirect)	Machinery and buildings (indirect)	Other	Total
Schader. 2009	Beef suckler cow farms - organic	LCA	GJ/ha	Cradle to gate	8.0	1.5	0.1	6.4	0.2	16.2
	Beef suckler cow farms - conventional	LCA	GJ/ha		11.2	3.9	2.2	7.8	0.4	25.5
Haas et al. 2001	Dairy - organic	LCA	GJ/tonne	Cradle to gate. Excluded energy in buildings/machinery	3.4	0.8	0.0	0.0	1.8	6.0
	Dairy - extensive conventional	LCA	GJ/tonne		4.1	3.7	0.2	0.0	0.6	8.6
	Dairy - intensive conventional	LCA	GJ/tonne		4.5	3.8	3.7	0.0	7.1	19.1
Venkat. 2012	Broccoli - organic	LCA	GJ/acre	Cradle to gate. Excluded energy in buildings/machinery	18.3	0.0	2.6	0.0	11.5	32.4
	Broccoli - conventional	LCA	GJ/acre		16.0	0.0	5.5	0.0	5.4	26.9
Karlen et al. 1995	Soybean, corn, oat, hay rotation, with manure charges, organic	Fam study	GJ/field	Cradle to gate includes labour but excluded energy in buildings and machinery	0.7	0.0	9.5	0.0	0.0	10.2
	Soybean, corn, oat, hay rotation, without manure charges, organic	Fam study	GJ/field		0.7	0.0	1.4	0.0	0.0	2.1
	Conventional corn and soybean	Fam study	GJ/field		0.8	0.0	3.2	0.0	0.3	4.3
Williams et al. 2006	Potatoes - organic	LCA	GJ/tonne	Cradle to gate	1.0	0.0	0.2	0.2	0.0	1.3
	Potatoes - conventional	LCA	GJ/tonne		0.8	0.0	0.4	0.1	0.0	1.3
	Bread wheat - organic	LCA	GJ/tonne		1.4	0.0	0.2	0.0	0.1	1.7
	Bread wheat - conv.	LCA	GJ/tonne		0.8	0.0	1.5	0.0	0.1	2.5
Alonso and Guzmán. 2010	Lettuce - greenhouse crop, organic	Farm surveys	GJ/ha	Cradle to gate, includes labour and embodied energy in machinery/buildings within 'other'	7.5	0.0	31.4	3.1	128.7	170.6
	Lettuce - greenhouse crop - conventional	Farm surveys	GJ/ha		4.7	0.0	3.5	3.1	140.1	151.4

4.7 Effect of functional unit when comparing studies

As found by Lynch *et al.* (2011), the unit of comparison affects the performance of organic farming systems with regard to environmental assessment criteria such as energy use. In common with this study, we have found that for most product types, organic performs better than conventional per unit of product, with over 75% of the product comparisons in Figure 4.2 reporting lower energy use. In particular, Figure 4.2 illustrates the efficiency of organic grazing systems, which is due to the lower energy impacts associated with forage production for beef and sheep production (organic energy-use ranges from 21 to 94% of conventional for these systems, depending on the system intensity). In common with Lynch *et al.* (2011), we have also found that organic systems tend to compare less favourably for poultry systems. Energy use under organic management was found to range from 125 to 160% of conventional for broilers. For egg production, energy use also tended to be higher, between 120 and 127% of the conventional barn and cage-based systems respectively. There was less difference between the energy requirements of organic and conventional free-range systems, with organic requiring 103 to 105% of the energy used on the conventional systems (Leinonen *et al.*, 2012b; Williams *et al.*, 2006).

With regard to crops, most organic systems perform better than conventional in energy use terms, mainly as a result of an absence of manufactured nitrogen fertiliser. Energy use for cereal cropping is approximately 80% of conventional per unit of product, despite the lower yield. Vegetable production energy requirements also tend to be lower on organic farms, requiring approximately 75% of the energy used under conventional. There are some exceptions, in particular glasshouse vegetables, apple and potato production exhibit reduced yields and similar levels of energy inputs, which can result in more energy use per unit of product overall. This is a result of greater losses from insect pests and diseases in the case of potatoes and apples. Reduced yields in organic glasshouse vegetable production systems were partly due to an increased occurrence of speciality cropping (e.g. vine tomatoes, Williams *et al.*, 2006).

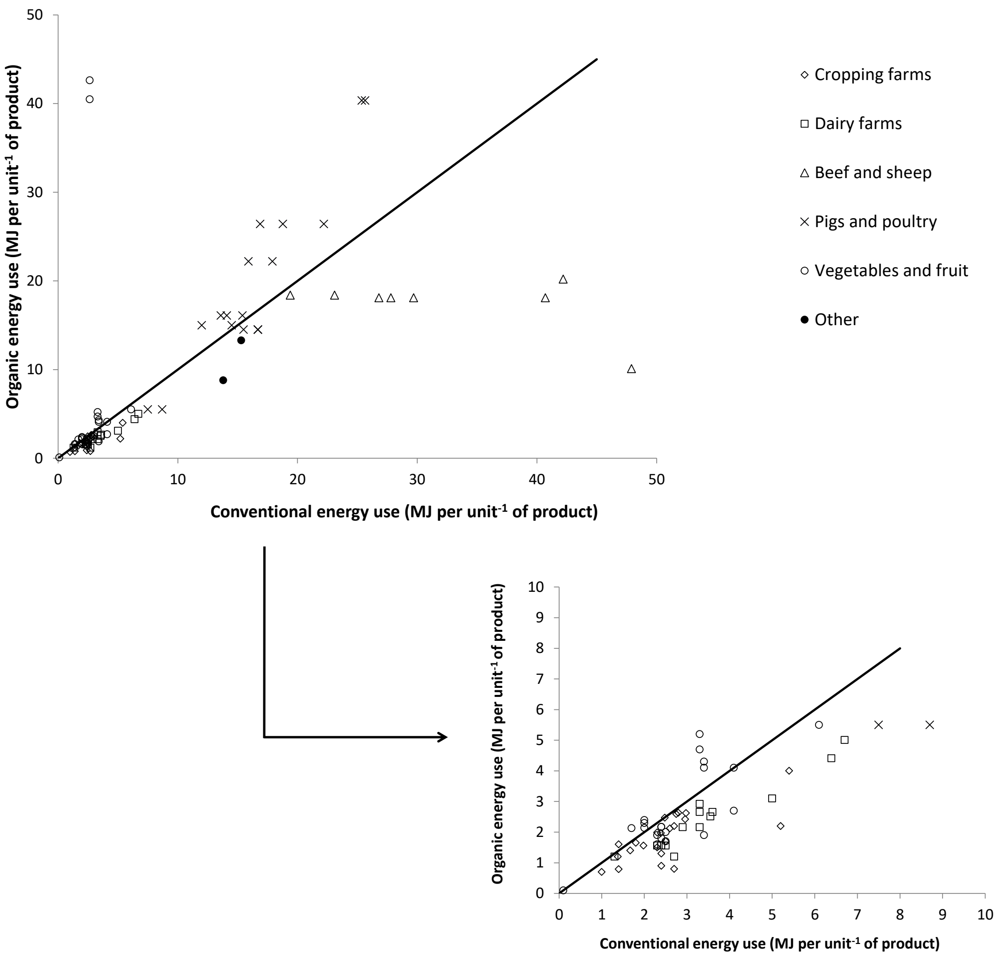

Figure 4.2: Organic vs conventional energy use per unit of product with expanded selection. Organic performs better below the line, worse above the line. Please note the 'trend-line' is x=y for the purposes of illustrating the relative performance for each product type and is *not* a line of best fit.

It is also clear from Figure 4.3 that the difference between conventional and organic systems is greater when comparisons are made on a per hectare basis, over 80% of the comparisons showing a lower energy use associated with organic production. This is to be expected due to the lower intensity of production on most organic holdings, resulting in fewer inputs, and a reduced yield. Despite this, organic performs less well when the energy content of the organic matter/compost used on organic holdings is taken into consideration. Average energy inputs per unit of land area were approximately double that of the conventional farms when this was taken into account (Alonso and Guzman; Karlen *et al.*, 1995). For the reasons outlined above, however, this renewable energy input cannot be compared in the same way to fossil-fuel based energy.

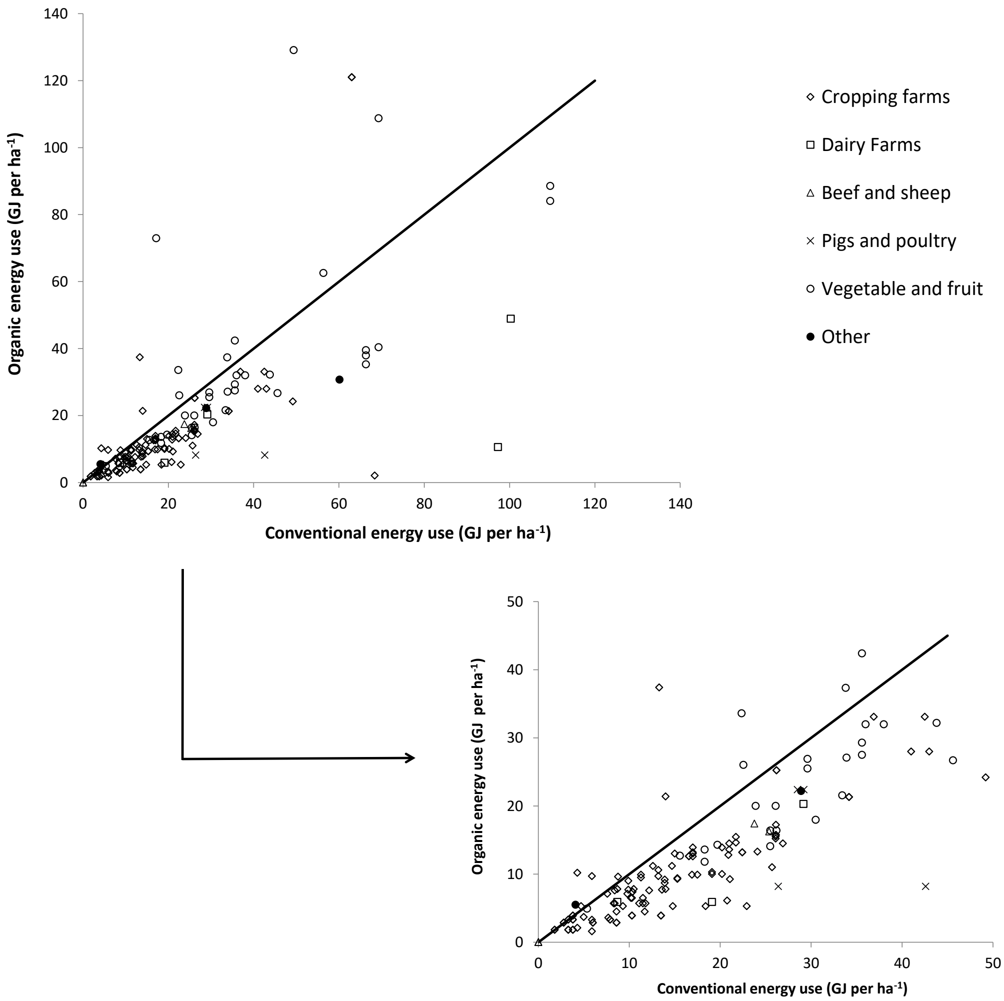

Figure 4.3: Organic vs conventional energy use per hectare with expanded selection. Organic performs better below the line, worse above the line. Please note the 'trend-line' is x=y for the purposes of illustrating the relative performance for each product type and is *not* a line of best fit.

A number of studies have compared organic and conventional systems in terms of energy efficiency (energy out/energy in). A range of approaches to measuring energy have been used with some authors expressing production of organic/non organic systems in terms of combustion energy (Pimentel *et al.*, 1983) and other authors using metabolisable energy output values (Cormack and Metcalfe, 2000). In addition some studies have included energy use associated with the production of farm infrastructure (e.g. buildings and machinery) whereas others have only focussed on energy use associated with feed, fertiliser and other variable inputs (Alonso and Guzman, 2010; Helander and Delin, 2004). Despite the variation in methods, it is possible to see that organic production outperforms conventional for nearly all of the products listed in Table 4.3. Again lower levels of inputs are the main reason for the increased efficiency of organic farming within these studies. There are some exceptions, however, for instance the Cormack and Metcalfe (2000) study found that the lower yield and the inclusion of fertility building crops within stockless arable farms resulted in a lower energy efficiency overall. Guzmán and Alonso (2008) also found that net efficiency is lower in organic olive production, mainly due to incorporated organic material originating from other ecosystems, although the organic systems performed better in terms of non-renewable energy use efficiency. Nguyen and Haynes (1995) also reported greater machinery use for weed control in organic pea production, which resulted in a lower energy efficiency overall, in a comparison of mixed farming systems in New Zealand.

Table 4.3: Energy ratios (energy output divided by input) for conventional and organic crops and livestock. All of the studies cited here contain statistical uncertainties; some authors have calculated these and others not, where individual values are presented these represent the average energy ratio. Ranges are presented where different treatments or sites have been used to compare the production systems, e.g. Nguyen and Haynes (1995).

Farm system or crop/livestock	Region of production	Org. OUT/IN	Conv. OUT/IN	Source
Crops				
Corn	USA	5.75-7.6	4.47	Pimental et al. (1983)
Spring wheat	USA	3.22-3.49	2.38	Pimental et al. (1983)
Potatoes	USA	1.07-1.20	1.28	Pimental et al. (1983)
Stockless arable farm	UK	4.41	5.18	Cormack and Metcalfe (2000)
Wheat	New Zealand	14.9-16.5	11.2-17.4	Nguyen and Haynes (1995)
Barley	New Zealand	15.4-17.5	9.9-16.3	Nguyen and Haynes (1995)
Peas	New Zealand	9.0-9.1	8.8-11.5	Nguyen and Haynes (1995)
Arable rotation	Canada	10.4	6.8	Hoeppner et al. (2006)
Arable and alfalfa rotation	Canada	33.5/11.9	19/7.4	Hoeppner et al. (2006)
Arable rotation: situation related pesticide use (2002-2006 experiment period)	Germany	17.4	20.7	Deike et al. (2008)
Arable rotation: reduced pesticide use (2002-2006 experiment period)	Germany	16.9	20.7	Deike et al. (2008)
Arable Farms	Sweden	4.3	5.9 Int: 5.9-6.5	Helander and Delin (2004)
Arable crops	Spain	1.88-8.27	4.52-6.7	Alonso and Guzmán (2010)
Vegetables and Fruits				
Vegetables	Spain	0.42-2	0.75-1.38	Alonso and Guzmán (2010)
Greenhouse vegetables (unheated)	Spain	0.13-0.22	0.21-0.28	Alonso and Guzmán (2010)
Irrigated fruits	Spain	1.73-5.89	4.88-5.48	Alonso and Guzmán (2010)
Rainfed fruits	Spain	1.33-2.82	1.87-2.14	Alonso and Guzmán (2010)
Apples	USA	1.18	1.11-1.13	Reganold (2001)
Apples	USA	0.06	0.89	Pimental et al. (1983)
Olive groves (without compost allocation)	Spain	2.4-5.2	2.2-4.2	Guzmán and Alonso (2008)
Olive groves (with compost allocation)	Spain	0.6-2.2	1.4-2.8	Guzmán and Alonso (2008)

Farm system or crop/livestock	Region of production	Org. OUT/IN	Conv. OUT/IN	Source
Livestock and mixed farms				
Sheep meat and wool	New Zealand	9.1-12	11-15	Nguyen and Haynes (1995)
Upland livestock (beef and sheep)	UK	2.47	1.1	Cormack and Metcalfe (2000)
Dairy	UK	1.67	0.43	Cormack and Metcalfe (2000)
Pig farms	France	1.59	1.17-1.20	van der Werf et al. (2007)
Mainly arable farm (some livestock)	UK	5.54	3.58	Cormack and Metcalfe (2000)
Experimental farms: Organic beef and arable and conventional croppping farm - higher value includes straw harvested	Germany	18/21	11.1	Kustermann et al. (2008)
28 commerical cropping and livestock/cropping only farms (18 organic and 10 conventional)	Germany	13 (6.0-19.3)	11.8 (6.1-16.2)	Kustermann et al. (2008)

4.8 Emergy studies

Most of the studies referred to above have concentrated on fossil fuel use when comparing the efficiency of organic and conventional systems. A limited number of studies have taken a different approach, using the emergy method to account for all energy inputs to the system, including human activity and ecosystem services (Bakshi, 2002; Odum, 1996). Emergy accounts for these inputs through an assessment of total amount of energy used for their creation. Scienceman (1989) therefore explains emergy as a calculation of the “energy memory” of systems (Brown and Herendeen, 1996). A common unit (i.e. solar emjoules – seJ) is used within emergy assessments, to express the amount of emergy required to produce a gram (seJ/g) or joule (seJ/J) of a particular resource, commodity or service. This is referred to as the ‘solar transformity’. The emergy-efficiency of different agricultural production systems can be compared through their relative solar transformities, with a lower transformity value per unit indicating a greater efficiency.

In addition to exploring the solar transformities of production systems, some emergy studies have investigated the Emergy Yield Ratio (EYR). This is an expression of the total emergy (in seJ) within a system in relation to the emergy purchased on the market (e.g. fossil fuels). In this sense, the EYR is a "measure of the systems net contribution to the economy beyond its own operation" (Odum, 1996). Other studies have also explored the Environmental Loading Ratio (ELR), which is the ratio of purchased and non-renewable local emergy to renewable environmental emergy. This measure can be used as an indicator of environmental stress and technological level (Odum, 1996). Emergy flow and emergy density are also used to explore levels of environmental stress through comparing the spatial and temporal concentration of emergy within different systems (e.g. emergy per unit of time or area, Castellini *et al.*, 2006).

Castellini *et al.* (2006) used the emergy approach to assess the efficiency of organic and conventional poultry production systems in Italy. Their study found that the solar transformity was lower within the organic system assessed, despite a lower level of production. This was due to the avoidance of chemical fertilisers and pesticides in the production of feed. In addition, the study found that the emergy costs for cleaning/sanitization of buildings were lower in the organic system, as a result of organic regulations only permitting molecules for sanitisation that have a low environmental impact. Through an assessment of the Energy Yield Ratio, the same study revealed a reduction in external inputs and in ecosystem stresses when under organic management. The organic system also had a higher use of renewable energy, as expressed through the ELR (see Table 4.4). In particular, this was through its reliance on organic sources of fertility (poultry and cow manure) as opposed to synthetic fertiliser. The emergy density within the conventional system was also approximately eight times higher than the organic, as a result of much greater use of non-renewable inputs.

Pizzigallo *et al.* (2008) also found a higher ELR for conventional systems of wine production in Tuscany, Italy, finding that the use of non-renewable resources on the conventional farm was approximately 15 times greater than that of renewable, while for the organic farm this level was only 10 times greater. The higher ELR for the conventional system was a result of the increased soil erosion and the use of manufactured fertilisers. Furthermore, the conventional system used a higher amount of agricultural machinery and fuel, plus a greater amount of glass for bottling (the organic farm used bottles that were lighter). The difference is thus not intrinsic to the farming system. The organic farm also had a lower solar transformity indicating a less resource intensive production system. However, the conventional farm was disadvantaged by a greater amount of on farm processing and the fact that only the best grapes were harvested (Pizzigallo *et al.*, 2008).

La Rosa *et al.* (2008) also used the emergy approach to compare organic and conventional red orange production from four Sicilian farms, this study also found a higher renewable energy use on the organic farm assessed, which contributed to a higher EYR and a much lower ELR. This was the result of a greater reliance on organic sources of fertility within the organic system, compared to the energy intensive manufactured fertiliser inputs used on the conventional farm. Furthermore, the conventional system used a greater amount of electricity per hectare. Conversely, the same study found a higher solar transformity (seJ/g) associated with two of the three organic farms assessed, as a result of the lower product yield.

In a comparison of wheat production in Denmark, Coppola *et al*. (2008) also found a lower emergy flow in organic production systems (i.e. lower seJ ha^{-1} $year^{-1}$) due to an absence of man-made fertilisers. Organic seed production was found to be more resource-intensive than conventional, and more field operations and greater machinery use were reported for the organic system. The study also reported a lower solar transformity for the organic wheat crop, suggesting a reduced efficiency per unit of biomass (straw and grain) despite the lower environmental impact, as expressed within the reduced ELR in Table

4.4. Ghaley and Porter (2013) also used the emergy method to compare two farming systems in Denmark; a conventional wheat production system and an organically managed Combined Food and Energy (CFE) system consisting of mixed arable cropping, clover ryegrass swards and woody biomass production. The emergy use in the conventional wheat system was 7.4 times higher than in the CFE, as a result of increased use of manufactured fertiliser and higher rates of soil erosion. The multiple yield components of the CFE system resulted in a greater output and a higher EYR. A lower ELR was also reported for the CFE system due to the reliance on renewable inputs (e.g. biologically fixed nitrogen). The study concludes that the CFE system provides a greater contribution to the economy compared with a wheat monoculture. The authors also suggest that such a diverse system could provide a suitable way forward for food and energy production, if an appropriate economic and policy environment could be provided.

Table 4.4: Results from five 'emergy' studies comparing organic and conventional production. Results expressed as 'seJ' = solar emergy joules or emjoules, i.e. units of solar energy that would be required to generate all the inputs to the farming system defined (expressed in seJ joule (j)$^{-1}$ or seJ gram (g)$^{-1}$ or seJ hectare (ha)$^{-1}$.

Author of study	Castellini et al. (2006)		Pizigallo et al. (2008)		La Rosa et al. (2008)		Coppola et al. (2009)		Ghaley and Porter (2013)	
Production system and country	Poultry (meat) production cycle: Italy		Wine production: Italy		Red orange production: Sicily		Wheat production: Denmark		Conventional wheat and organic Combined Food and Energy (CFE) system: Denmark	
System type – conventional / organic	Conv.	Org.	Conv.	Org.	Conv.	Org.	Conv.	Org.	Conv. (wheat)	Org. (CFE)
Solar transformity (seJ J^{-1})	6.1×10^5	5.7×10^5	N/A	N/A	N/A	N/A	$3.9\text{-} 5.8 \times 10^4$	$4.6\text{-}7.1 \times 10^4$	8.63×10^4	6.40×10^3
Solar transformity (seJ g^{-1})	4.3×10^9	4.1×10^9	4.7×10^{15} (per tonne)	3.0×10^{15} (per tonne)	1.2×10^9	$0.6\text{-}2.2 \times 10^9$	N/A	N/A	N/A	N/A
Emergy flow (seJ ha^{-1})	N/A	N/A	N/A	N/A	N/A	N/A	$6.6\text{-}6.9 \times 10^{15}$	$5.4\text{-}5.6 \times 10^{15}$	N/A	N/A
Empower density (seJ year^{-1}) m^2	7.8×10^{14}	3.6×10^{12}	1.9×10^{16}	1.0×10^{16}	N/A	N/A	N/A	N/A	N/A	N/A
Emergy Yield Ratio (EYR)	1.1	1.5	N/A	N/A	1.5	1.6-11.7	N/A	N/A	1.0	1.2
Environmental Loading Ratio (ELR)	5.2	2.0	15.4	10.5	43.0	3.8-30	7.3-8.5	2.3-2.4	37.7	4.2

Emergy is clearly a useful method that presents a more complete picture of the energy and ecosystem costs and benefits associated with a range of farming systems (Gomiero *et al.*, 2008). Unlike energy accounting, the emergy approach allows for an assessment of a productive system's relationship with the environment, in terms of energy flows. It takes into account environmental inputs that are usually treated as 'free' (e.g. ecosystem services, Bakshi, 2002; Pizzigallo *et al.*, 2008), assessing the amount of natural 'labour' required to obtain a given product (Castellini *et al.*, 2008). Despite these perceived advantages, the emergy approach has been criticised on the basis of the subjective judgements and associations that lead to the allocation of solar energy values to inputs such as wind and rain (Jones, 1989). The lack of a sufficiently detailed explanation behind the underlying methodology within many of the calculated solar transformities has contributed to this criticism, (Hau and Bakshi, 2004) although recent attempts have been made to apply uncertainty calculations to the emergy approach (Li *et al.*, 2011). Hulsbergen *et al.* (2001) also state that inclusion of solar radiation in the energy balance can mask the variation of fossil energy input influenced by different husbandry techniques, as fossil energy is often a small proportion of the total emergy use when considering solar inputs. Conversely, it can also be misleading to focus only on the use of energy on-farm (i.e. without accounting for the embodied energy associated with inputs and natural services) providing an advantage to farms dependent on external sources for the maintenance of higher levels of production (Topp *et al.*, 2007). It has been suggested that a combined approach of using LCA and emergy analysis may help both methods to improve, allowing LCA to account for ecosystem services, and overcoming problems with allocation (i.e. partitioning energy inputs between multiple outputs) found within the emergy approach. This combined method was adopted by Pizzigallo *et al*. (2008), who used LCA methods to comprehend and disaggregate the productive systems assessed, together with the application of emergy to account for the energy contribution of ecosystems.

4.9 Discussion

4.9.1 Comparisons by farming system

When making comparisons of the energy efficiency of organic and conventional systems, it is difficult to draw definitive conclusions, partly as a result of the variation within each of the sectors, which makes performance very site and system dependent (Seufert *et al.*, 2012). For example, Williams *et al.* (2010) found that wheat grown on sandy soils uses about 20% more energy than on clay soils, within an LCA of organic and conventional arable crops grown in the UK. Refsgaard *et al.* (1998) also found that differences in soil type had a greater effect on energy efficiency than organic or conventional farming practices. Nevertheless, in common with the findings of Lampkin (2007), Lynch *et al.* (2011) and Gomiero *et al.* (2008) it is possible to state that for most grazing systems, organic farming will result in a lower energy use, on a unit area or weight of product basis. 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 the conventional practice (Deike *et al.*, 2008; Hoeppner *et al.*, 2006; Küstermann *et al.*, 2008). Similarly, for dairy systems, organic production tends to result in lower energy use per litre of milk produced, due to greater energy efficiency in the production of forage and reduced reliance on imported concentrates (Cederberg and Mattsson, 2000; Haas *et al.*, 2001; Thomassen *et al.*, 2008). With regard to poultry, meat and egg production tends to require more energy per kilogram of product and per hectare under organic management, as poorer overall feed conversion ratios and higher mortality rates reduce overall efficiency (Leinonen *et al.*, 2012a; Williams *et al.*, 2006).

With regard to cropping systems, the absence of fertiliser inputs tends to more than compensate for a lower yield within organic cereal production, resulting in lower energy use per kilogram of product (Pelletier *et al.*, 2008; Williams *et al.*, 2006) or little difference overall (Nemecek *et al.*, 2005). Organic management can also be better in terms of energy use for field vegetable production, as a result of fewer inputs in manufactured fertilisers and herbicides, although in some cases the energy used for flame weeding can make it worse (Cormack and Metcalfe, 2000). For organically produced potatoes, energy use tends to be greater due to yield losses from pests, causing lower yields overall (Williams *et al.*, 2006). Pimental *et al.* (1983) found that organic potato yields were only 50% of conventional as a result of a lack of control of blight (*Phytopthora infestans*) resulting in much lower energy efficiency per kilogram of product.

With regard to on farm energy use, in common with the study by Lynch *et al.* (2011) this review has found that in many cases organic farmers' diesel requirements are comparable to conventional; although for some crops this energy use may be greater through increased reliance on mechanical tillage, e.g. for broccoli (Venkat, 2012), wheat and potatoes (Williams *et al.*, 2006). The reduced tillage systems commonly found on conventional farms will also require less diesel than the 'traditional' mouldboard ploughing technique commonly used on organic farms, although the difference may be offset by indirect energy, depending on the rate/efficiency of usage (Clements *et al.*, 1995; Robertson *et al.*, 2000). With regard to indoor crops, a greater amount of energy is used for greenhouse production under organic management on a kilogram of product basis, as a result of lower yields but similar energy requirements for heating or building construction (Alonso and Guzman, 2010; Williams *et al.*, 2006).

The 'human energy' aspect is missing from many of the studies considered here. This is a result of the absence of a widely accepted and applied methodology for its inclusion, in addition to the relatively small contribution of labour to total energy use in modern cropping systems. Borin *et al.* (1997), for example, calculated that this aspect accounts for less than 0.2% of the total energy input in modern cropping systems. Relatively high energy input is likely, however, in other systems, such as fruit, vegetable and livestock production. The limited number of studies that have included this aspect found that organic farming will generally result in greater levels of on farm energy from human labour (Karlen *et al.*, 1995; Nguyen and Haynes, 1995; Pimentel *et al.*, 1983). Although this may have negative effects on the productivity per labour hour, some authors have taken an optimistic view of the increased labour requirements associated with organic production systems. For instance, Pretty (1998) in Cobb *et al*. (1999) found that a shift towards an organic production scenario in the UK could create 100,000 jobs in addition to encouraging more added value through on farm processing of products and direct sales.

4.9.2 Productivity vs. energy efficiency

It is also important to note that most of the studies and farming systems mentioned above found higher levels of productivity in conventional systems, despite organic systems having greater resource use efficiency. In this context Deike *et al.* (2008) point out the large yield losses that would result from a widespread switch to organic production. The lower yields from organic management have led some authors to conclude that organic farming is incapable of feeding the world in a sustainable manner (Connor, 2008; Trewavas, 2004). Others have claimed that the apparent benefits of organic production such as reduced fertiliser manufacture and pesticide use are a poor exchange for a potential lack of productivity (Powlson *et al.*, 2011). Despite this, a recent meta-analysis by Seufert *et al.* (2012) found that under good management practices, some organically grown food crops can nearly match conventional yields. Specifically, organically produced legumes and perennials on rain fed, weak acidic to alkaline soils were found to have small yield differences of less than 5%, although the authors of this study note the small sample size and high uncertainty for these crops. On the other hand, for vegetables and cereals, a greater, statistically significant yield reduction was found for organic systems (-33% and -26% respectively). The authors note that when only the most comparable organic and conventional systems are used, organic yields can be up to 34% lower. Conversely, a study based at the Rodale Institute’s experimental farm in the north eastern United States demonstrated that under drought conditions, crops in organically managed systems can produce higher yields than conventional crops. Yield increases within this study ranged from 137% to 196% of conventional depending on the crop and method of fertilisation (Lotter *et al.*, 2003). The main reason given is the increased water holding capacity of the soil, as a result of an increased organic matter content. Smolik *et al.* (1995) also found that yields within an organic system were more stable in the face of diseases and weather variation over a six year period.

Whatever the yield differences between organic and conventional production, it is clear from both an environmental and economic perspective that we need to reduce our reliance on fossil fuels, per unit of food produced, whether under an organic or conventional production scenario. Although the use of these reserves has clearly had a positive impact in terms of increasing productivity throughout the 'Green Revolution' (Godfray *et al.*, 2010) and fertiliser manufacture efficiency is increasing (Woods *et al.*, 2010) it has been highlighted that oil and gas reserves are only sufficient to meet our needs for another 50 to 100 years (Crews and Peoples, 2004). Moreover, the negative effects of our dependency on non-renewable inputs are already being witnessed (e.g. through food price riots in 2008, in part caused by increasing costs of fertiliser and fuel, Piesse and Thirtle, 2009). The wisdom of putting our faith in the development of an unproven or unknown energy source to maintain or increase levels of production in the future has also been questioned (Crews and Peoples, 2004). In addition recent assessments have found that vast increases in yield seen in recent years have been at the expense of increases in soil erosion, reductions in biodiversity and a large increase in agriculture's reliance on manufactured fertilisers and pesticides (Millennium Ecosystem Assessment, 2005; Tilman, 1999). In this context Gomiero *et al.* (2008) highlight the usefulness of methods such as emergy accounting, which can present a more complete picture of agricultural systems' impact on the natural environment. The current application of emergy approaches to comparisons of organic and conventional farming systems has been limited, however, and more work comparing the two approaches using this method would be helpful.

It should also be noted that in their current form, organic systems do not offer a radical alternative to the fossil fuel reliance of modern agricultural systems. The reduced use of energy in organic production and increased energy efficiency compared to conventional production is often marginal. These systems often still depend on the same sources of (fossil fuel) for tractors, machinery and buildings etc. While organic production can make a contribution to a more resource efficient agriculture, in its present form it does not provide a complete solution.

Some have suggested that a ‘happy medium’ for the development of more fossil fuel efficient farming systems would be to pursue lower input conventional farming systems (e.g. by reducing man-made fertiliser inputs, increased use of legumes for nitrogen fixation and organic manures, Foresight, 2011). Indeed recent work has highlighted that well managed conventional systems with reduced input levels can outperform organic production in terms of resource use efficiency, when measured on an energy output/input basis (Tuomisto *et al.*, 2012a). In this context, the recent International Assessment of Agricultural Knowledge, Science and Technology for Development (McIntyre *et al.*, 2008) and Foresight (2011) reports outline a number of key challenges to maintain the production of food whilst decreasing dependence on fossil energy, none of which would seem to exclude or preclude a conversion to organic standards:

- The development of decentralised, locally based production and distribution systems
- Improving nutrient use, in particular more exact timings and amounts of fertilisers (organic and inorganic)
- Increasing productivity through increasing the marketable/edible yield from crops, improved animal breeding, feeding and pest and disease control
- Recycling of urban and industrial waste
- Increased use of renewable energy throughout the supply chain

In addition, the need to improve the synchrony between N supplied by legumes and N demand from crops is highlighted by Myers *et al.* (1997). However, even with developments in this area, it will be difficult to match the synchronisation with crop demand to the same extent as through targeted application of soluble nitrogen through manufactured fertiliser (Cassman *et al.*, 2002; Crews and

Peoples, 2004). Crews and Peoples (2004) also highlight the importance of reducing the amount of grain fed to livestock, thereby freeing up land for legumes and reducing agriculture's current dependence on manufactured fertiliser. This would, however, particularly reduce the output of eggs and poultry meat and, to a lesser extent, pig meat, given the nutritional requirements of these stock. Kumm (2002) also highlights the importance of focussing meat production on landscapes that cannot be used for arable cropping, and using by-products that can contribute to food supply only through the refinement of meat producing animals. Although Kumm (2002) also highlights that in situations of energy shortage, there might be competition between meat production and the bioenergy sector.

4.10 Conclusion

Organic production systems focus on the development of closed cycles of production as far as this is possible, as espoused by the IFOAM principles. This naturally creates systems which are less productive. Results from studies considered within this review, however, have illustrated that the reduced yields are matched by greater energy efficiencies for most ruminant livestock and field crops. The difference is greatest when comparisons are made on a unit of area basis, although substantial increases in energy efficiency can also be observed per unit of product within most of the comparative studies. The difference between organic and conventional production tends to be greatest for grassland systems, due to the relative efficiency of producing grass in conjunction with clover, a practice encouraged within the organic sector. There are some important exceptions where organic performs worse. For example potatoes where a lower yield reduces efficiency and other vegetables that require flame weeding. Within livestock production, organic pig and poultry production systems also perform worse where poor feed conversion and higher mortality rates can lead to lower energy efficiency overall. The limited number of emergy analyses comparing the two production systems to date have also found a lower environmental loading and increased renewable energy use on organic farms. Overall it would appear that the energy efficiency of most cropping and ruminant livestock farming systems can be enhanced through the adoption of organic management. However, in many cases this will be at the expense of crop or livestock yields.

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CHAPTER 5. ASSESSING THE SUSTAINABILITY OF ORGANIC CROP ROTATIONS

Article title	Predicting the effect of rotation design on N, P, K balances on organic farms using the NDICEA model
Journal and publication status	*Renewable Agriculture and Food Systems.* Published
Co – authors	Davide Tarsitano, Cairistiona Topp, Stephanie Jones, Christine Watson (Scotland's Rural College) Adrian Williams (Cranfield University), Catherine Gerrard, Bruce Pearce (The Organic Research Centre)
Co-author contributions	Data provision from Scotland's Rural College's long-term experimental sites. Comments on draft text
Research methods applied	Data extraction and analysis to determine typical organic rotations and their representativeness Model validation and error assessments (Root Mean Square Error and Standard Error calculations) Modelling of site-specific N availability under contrasting soil / rainfall conditions

5.1 Abstract

The dynamic model Nitrogen Dynamics in Crop rotations in Ecological Agriculture (NDICEA) was used to assess the nitrogen (N), phosphorus (P) and potassium (K) balance of long-term organic cropping trials and typical organic crop rotations on a range of soil types and rainfall zones in the UK. The measurements of soil N taken at each of the organic trial sites were also used to assess the performance of NDICEA. The modelled outputs compared well to recorded soil N levels, with relatively small error margins. NDICEA therefore seems to be a useful tool for UK organic farmers. The modelling of typical organic rotations has shown that positive N balances can be achieved, although negative N balances can occur under high rainfall conditions and on lighter soil types as a result of leaching. The analysis and modelling also showed that some organic cropping systems rely on imported sources of P and K to maintain an adequate balance and large deficits of both nutrients are apparent in stockless systems. Although the K deficits could be addressed through the buffering capacity of minerals, the amount available for crop uptake will depend on the type and amount of minerals present, current cropping and fertilisation practices and the climatic environment. A P deficit represents a more fundamental problem for the maintenance of crop yields and the organic sector currently relies on mined sources of P which represents a fundamental conflict with the International Federation of Organic Agriculture Movements organic principles.

5.2 Introduction

Organic cropping systems focus on feeding the soil, rather than the plant, to build long-term system health and resilience (Lampkin, 2002). This approach results in a reliance on fertility building ley periods and the application of composts and manures, which supply a source of nutrition for the growing crops, whilst potentially improving the soil microbial life and organic matter contents (Lampkin, 2002; Watson *et al.*, 2002). The length of the ley period can vary from short term (12-18 months) to long-term (around 5 years), but typically the ley is kept for about 18 months to 3 years. In Europe, organic farmers most frequently use grass-clover mixes for their leys, with white clover *(Trifolium repens*) and red clover (*T. pratense*) being popular legume species and perennial ryegrass (*Lolium perenne*) and Italian ryegrass (*L. multiflorum*) as commonly chosen grass species (Döring *et al.*, 2013). The crops following the ley period make use of the built-up fertility, although the ley period can also remove fertility, in particular K in conserved grass (silage). Cropping following the ley phase often includes rotational use of over-winter green manures and cover crops such as cereal rye (*Secale cereale*) and vetch (*Vicia satvia*) to reduce losses and to supply additional N through biological fixation (Lampkin, 2002).

The use of these approaches on organic farms creates systems in which the nitrogen supplied is in a less available form, compared with conventional systems using mineral fertiliser (Torstensson *et al.*, 2006). The supply of available nitrogen in organic systems can therefore be a limiting factor for the maintenance of crop yields (Berry *et al.*, 2002; Dawson *et al.*, 2008; de Ponti *et al.*, 2012). In addition, poor synchronicity between the supply and demand for nitrogen can lead to leaching and gaseous losses, particularly following ley cultivation. Nevertheless, this is also an issue for conventional farmers, particularly following periods of high rainfall (Dawson *et al.*, 2008; Liang *et al.*, 2011; Patil *et al.*, 2010). Under organic management, the surplus of N following ley cultivation can be followed by an N deficit later in the crop rotation (Berry *et al.*, 2002). Although this shortage can be resolved through the application of organic composts, manures, and/or through the use of short-term green manures, it can be difficult to match the N supplied from such sources with crop demand. A reliance on such methods can therefore contribute to lower nitrogen use efficiencies compared to non-organic systems applying targeted mineral N (Dawson *et al.*, 2008; Torstensson *et al.*, 2006). Despite the challenges of N availability and synchronicity of N supply/demand on organic farms, Berry *et al.* (2003) found positive N balances in a comparison of nine organic farms, and reported that the farms were probably sustainable in terms of N supply and offtake. However, the same study found that phosphorus (P) and potassium (K) levels were in deficit within the stockless systems assessed, and that only farms with large manure returns from stock fed with bought-in feed had a positive or neutral K budget. Korsaeth *et al.* (2012) and Torstensson *et al.* (2006) also found P and K deficits within organic arable cropping and mixed dairy farming systems in Norway and Sweden respectively.

The research presented here aimed to assess the effect of rotation design on the supply and offtake of nitrogen (N), phosphorus (P) and potassium (K) on organic farms using the dynamic model NDICEA (Nitrogen Dynamics In Crop rotations in Ecological Agriculture). Five hypotheses were posed at the outset of this study. First, NDICEA can effectively calculate the course of mineral-N over a range of organic crop rotations. Second, the N supplied through biological fixation in stockless organic rotations is sufficient to support crop offtake. Third, organic cropping systems incorporating livestock manure applications are able to maintain a positive or neutral N, P and K balance. Fourth, organic rotations will typically rely on imported P to maintain a balance of this nutrient. Fifth, that a deficit of K is a common feature in the overall nutrient balance of typical organic crop rotations.

5.3 Methods

The NDICEA model (Van der Burgt *et al.*, 2006) was applied to assess the supply and demand of nitrogen (N), phosphorus (P) and potassium (K) within a range of stocked (i.e. with manure) and stockless rotations applied at experimental organic farms in the UK. In addition, typical organic rotations were drawn from the literature.

5.4 Model description

NDICEA is a dynamic, target-oriented model with crop yield and crop quality parameters, e.g. dry matter, N, P and K contents, used as a basis for crop uptake calculations. Mineralisation of nitrogen from soil organic matter (SOM) and organic inputs such as manure and compost is also calculated, factoring in the effects of weather, irrigation and soil type, although the model does not account for volatilisation losses during composting / storage of manure. The model uses a daily time step, utilising site specific weather data (rainfall, temperature, evapotranspiration) and user-defined soil and crop parameters. Although the model contains default values for a range of soil types these values can be automatically adjusted through the addition of data on measured soil mineral nitrogen and SOM within the user interface. Following the entry of these values calibration of the model takes place through the implementation of an algorithm that selects an optimum parameter value from a range of plausible values for such variables as N leaching, denitrification and water holding capacity (Van der Burgt *et al.*, 2006). Within this study measured values of soil mineral nitrogen and SOM were used to calibrate the model runs and improve the accuracy of the assessments.

A repeat calculations function within the model also allows the user to assess the longer term impacts of rotations both in terms of the nutrient supply and the effect on organic matter stocks. The focus of the model is on nitrogen dynamics. For P and K, a simpler farm-gate balance approach is taken (i.e. only crop offtake and atmospheric deposition is calculated, based on the user-defined input parameters and/or default values). The calculations for P and K are also unaffected by changing the soil type or daily rainfall and evapotranspiration values within the model. The wide range of cover crops and green manure-options within the NDICEA interface makes the tool particularly applicable for organic farmers, however the tool can also be used to improve understanding of nitrogen dynamics under non-organic management (Van der Burgt *et al.*, 2006). Under both organic and non-organic management model performance will be improved by calibration, with a higher number of measurements improving the accuracy of the estimates of N supply and losses (van der Burgt and Rietberg, 2012).

5.5 Description of sites and cropping systems

The model was run using crop, soil and weather data from the UK Government funded organic conversion trials held at ADAS Terrington (Cormack, 2006), Warwick University's Hunts Mill site (Lennartsson, 2000) as well as other long-term trials at Elm Farm Research Centre (EFRC, Welsh *et al.*, 2002), Scotland's Rural College (SRUC Tulloch and Woodside, Taylor *et al.*, 2006) and a grazing only trial at the Institute of Biological, Environmental and Rural Sciences (IBERS) at the University of Aberystwyth (Ty Gwyn, Haggar and Padel, 1996). Please see Table 5.1 and Figure 5.1 for more information on the trials.

Soils data from each site were collected from project reports, site records and published literature (Cormack, 2006; Haggar and Padel, 1996; Lennartsson, 2000; Taylor *et al.*, 2006; Welsh *et al.*, 2002). The bulked soil samples at each site were taken along a W transect twice each year in the case of Elm Farm (Welsh *et al.*, 2002), Warwick University (Lennartsson, 2000) and ADAS Terrington (Cormack, 2006, after sowing and harvest) and once per year at the SRUC sites (Watson *et al.*, 2011) and at Ty Gwyn (Haggar and Padel, 1996, January and April respectively). Samples were analysed for available P (Modified Morgan's solution at SRUC sites and Olsen's method at other sites), available K (Modified Morgan's solution at SRUC sites and ammonium nitrate extraction at ADAS and Elm Farm), mineral nitrogen (potassium chloride solution) and organic matter (loss on ignition).

Table 5.1: Crop Rotations used at each of the experimental sites (C = Carrots, G/C = Grass White Clover, P = Potatoes, RC = Red clover, SBA = Spring barley, SB = Spring beans, SO = Spring oats, S = Swede, SW = Spring wheat, WB = Winter beans, WO = Winter oats, WW = Winter wheat):

	Rotation year									
Rotation	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
EFRC A	RC	WW	WW	SO						
EFRC B	RC	P	WW	WO						
EFRC C	RC	WW	WB	WW						
ADAS Terrington	RC	RC	P	WW	SB	SW	RC	P	WW	SB
Warwick, Hunts Mill - Area 1	SB	G/C	G/C	P	C	SBA	G/C			
Warwick, Hunts Mill - Area 6	P	C	SB	P	C	SBA	G/C			
SRUC - Tulloch T50	G/C	G/C	G/C	SO	S	SO				
SRUC - Tulloch T67	G/C	G/C	G/C	G/C	SO	SO				
SRUC - Woodside W37	G/C	G/C	SO	P	SO	G/RC	S	SO		
SRUC- Woodside W50	G/C	G/C	G/C	SO	P	SO				
IBERS - Ty Gwyn	G/C	G/C	G/C							

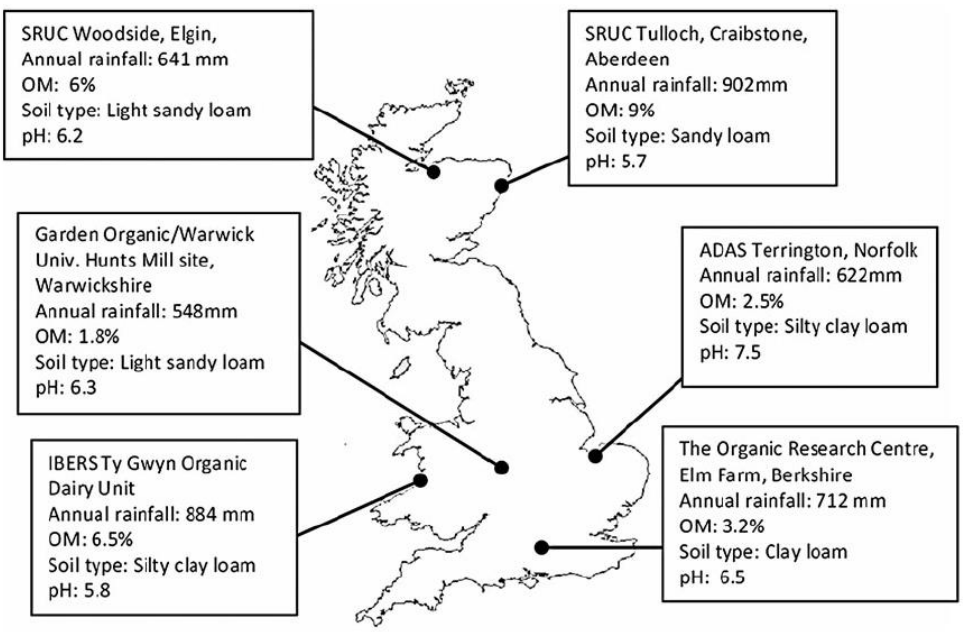

Figure 5.1. Approximate location and site parameters for each of the long-term organic trial sites. OM = organic matter content of the soil (% loss on ignition). Rainfall amounts are mean values over the course of the trial(s).

Soil samples were taken at a range of depths. At Elm Farm separate topsoil (0-15cm) and subsoil (15-30cm) samples were assessed for the above parameters. At Warwick University assessments were carried out on samples from 0-30cm and 30-60cms. At Ty Gwyn mineral nitrogen was sampled to 80cm in 15cm and 20cm increments respectively, although only the first sample layer was assessed for P, K and organic matter. At ADAS Terrington all samples were taken to 90cm in 15cm increments. At Woodside, the mineral nitrogen was sampled to 45cm in increments of 15cm, and at Tulloch, the mineral nitrogen was sampled to 30cm in increments of 15cm.

The rotations applied at EFRC and ADAS Terrington were managed as stockless systems, although phosphate fertilisers permitted under organic standards were applied. The EFRC trial received lime up to a maximum rate of 2 t ha^{-1} yr^{-1}. Lime was similarly applied at the ADAS site in order to keep the pH between 6 and 6.5. All of the red clover leys at each site were managed through cutting and mulching. The Hunts Mill plot trials included both 'with manure' and 'without manure' treatments. Both sites at Hunts Mill received a single application of green waste compost at a rate of 20 tonnes per hectare. At the Ty Gywn organic dairy unit, manure was deposited at a rate consistent with 2 Livestock Units (LU) per hectare and lime was applied at a mean rate of 0.7 t ha^{-1} over 3 years. At both of the SRUC sites (i.e. Tulloch and Woodside) total annual manure applications were based on 2.8 LU ha^{-1} for the period 1991-1998. In addition ground limestone and potassium sulphate (K_2SO_4) were applied to all Woodside plots in 1991, at a rate of 3.75 t ha^{-1} and 150 kg ha^{-1} respectively. All grass-clover leys at SRUC sites were managed through a combination of grazing with sheep and cutting for silage as described in Taylor *et al.* (2006).

In addition to data from the organic trials, information on typical organic rotations was gathered, based on examples within the Organic Farm Management Handbook (Lampkin *et al.*, 2011b) and following guidelines given to organic farmers with respect to the proportion of fertility building leys to exploitative phase (see Table 4.3 for description of the rotations used, Lampkin, 2002; Lampkin *et al.*, 2011b). Manure application rates for the typical stocked cropping systems were derived using typical livestock numbers for cropping farms (i.e. 0.3 Grazing Livestock Units per Utilisable Agricultural hectare) reported within a sample of approximately 30 organic farms included within the FBS (Farm Business Survey) for England and Wales (Moakes and Lampkin, 2010; Moakes and Lampkin, 2011; Moakes *et al.*, 2012a). Rock phosphate application rates were derived with expert input from the Institute of Organic Training and Advice (IOTA) registered advisors.

Table 5.2: Typical organic rotations assessed within this study (G/WC = Grass/ white clover, RC/G = Red clover/grass, SO - Spring oats, SW = Spring wheat, SB = Spring beans, WO = Winter oats, WW = Winter wheat, WR = Winter rye, P = Potatoes, PE = Peas)

	Rotation year									
Rotation	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Stocked 'complex'	G/WC	G/WC	G/WC	WW	WO	RC/G	RC/G	P	SB	SW
Stocked 'simple'	RC/G	RC/G	WW	P	WW	WR				
Stockless 'complex'	RC/G	RC/G	P	WO	SB	SW				
Stockless 'simple'	RC/G	WW	PE	SO						

The rotations were chosen to represent a range of stocked and stockless organic cropping systems. To assess the representativeness of the rotations crop areas were compared to those reported for a stratified sample of 30 organic farms included within the Organic Farm Income Reports published by Aberystwyth University and The Organic Research Centre (Moakes and Lampkin, 2010; Moakes and Lampkin, 2011; Moakes *et al.*, 2012).

As shown in Figure 5.2, the typical rotations are broadly representative of the crop areas reported on actual organic farms within the Farm Income Reports' matched sample. Although there are some differences by crop type (e.g. both stocked simple and stockless simple containing a high percentage of cereal crops) the differences are generally in the region of 15-20%. In view of the wide variation between the rotations on individual farms, this is an acceptable margin of error and the rotations applied here can be considered to be broadly representative of organic cropping farms.

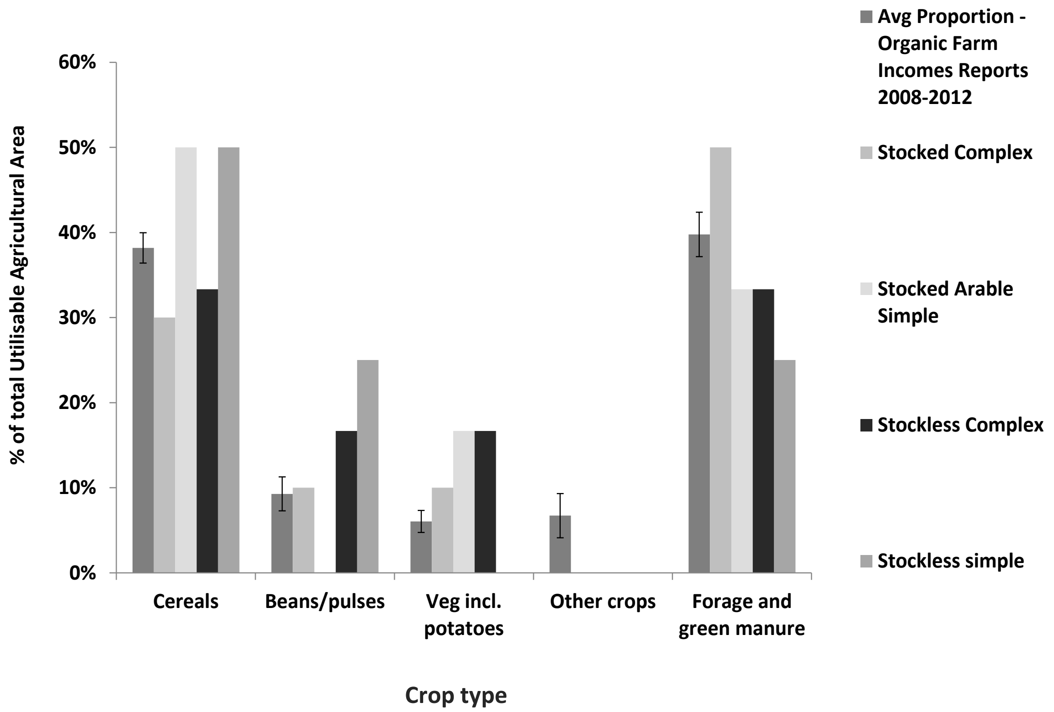

Figure 5.2: Comparison of the land use by crop type for the typical rotations used within this study to data for 30 'Cropping Farms' collected with the FBS-based Organic Farm Incomes Reports (2010-2012). Error bars = standard error.

5.6 Model application

The model was applied to assess the effect of rotation design on the supply and offtake of nitrogen (N), phosphorus (P) and potassium (K) on the above experimental sites and within typical organic rotations. The measured changes in SOM over time were small for most of the sites assessed (data not shown). It was therefore necessary to run the model a number of times to ensure a minimal gain or loss of SOM and to avoid erroneous conclusions. Based on measured data and results from long-term experiments (Johnston *et al.*, 2009; Melillo *et al.*, 1989) a uniform, near steady-state was assumed to have been reached once the annual change in SOM was less than 2% of the total organic matter pool (expressed in kg ha^{-1}) over the rotation. Two runs of the model were implemented for each site, an uncalibrated run, using only the basic, user-adjusted soil and crop parameters, and a calibrated run following the input of measured amounts of soil mineral N and SOM to make the model automatically adjust advanced soil parameters such as N leaching and denitrification factors. The root mean square errors (RMSE) were calculated based on the size of the deviation between the measured soil N and modelled soil N values and the number of samples at each site. The observed N values were used therefore in the calibrated runs as both inputs to the model and as comparator for assessing model performance. Following the calculation of N, P and K balances for the trial sites, a further application of the model was implemented for the typical organic rotations desribed above, using the same soil and weather conditions as the trial sites.

5.7 Results and discussion

The modelled estimates for Soil Mineral Nitrogen (SMN) were compared with the sampled soil N values from each of the sites to test the ability of the model to simulate the measured rotations. An NPK balance for each of the rotations was then calculated.

5.7.1 Comparison of the NDICEA model's estimates for soil N with measured mineral N values at each of the trial sites

A RMSE of 20 kg N ha^{-1} or less was proposed by Van der Burgt *et al.* (2006) to represent acceptable model performance for practical purposes. This could be achieved for most of the sites, although in some cases (e.g. EFRC and Hunts Mill) the modelled results are above this value (see calibrated model output in Table 5.3). The higher errors at EFRC could be a result of the small number of measurements (i.e. 7). The high errors for the stocked rotations at Hunts Mill could be explained by the fact that soil N measurements were taken soon after application of manures (the modelled values for this phase of the rotation were more than 100 kg N ha^{-1} lower than the recorded values). It may also be possible that the model is underestimating the nitrogen supplied or that the deeper samples at Hunts Mill (0-60cm) resulted in a mixing of topsoil and subsoil layers, and a subsequent overestimate of the mineral N content in the subsoil layer.

Table 5.3: Comparison of the error found in calculating soil N (kg N ha^{-1}) produced by uncalibrated and calibrated runs of NDICEA using data collected for the rotations applied at each experimental site (RMSE = Root Mean Square Error across all measurements, n = number of soil mineral N samples used for calibration of the model)

Site / experiment	n	RMSE uncalibrated model	RMSE calibrated model
EFRC A	7	48.8	16.6
EFRC B	7	57.6	48.7
EFRC C	7	30.4	21.5
ADAS Terrington	10	10.9	6.3
Warwick, Hunts Mill Area 1 with FYM	12	47.7	44.1
Warwick, Hunts Mill Area 6 with FYM	12	22.3	19.3
Warwick, Hunts Mill Area 1 no FYM	12	41.0	38.8
Warwick, Hunts Mill Area 6 no FYM	12	23.7	18.6
SRUC Woodside W37	30	12.8	11.6
SRUC Woodside W50	30	14.1	11.8
SRUC Tulloch T50	5	22.0	18.9
SRUC Tulloch T67	5	13.4	6.0
Ty Gwyn	2	8.3	7.5

5.7.2 NPK balance for each of the rotations applied at the organic trial sites

Modelled nutrient balances derived from NDICEA are presented in Table 5.4 for each of the stockless rotations. The results include an estimate of the change in SOM, with a negative number indicating mining of existing reserves and a positive number indicating an assimilation of N to SOM (i.e. an increase in organic matter stocks).

Table 4.4 illustrates that the amount of N supplied through biological fixation could potentially support the crop removal at the EFRC trial, although due to losses from the system, in particular leaching, much of this N is lost. The negative values for organic N indicated a mining of organic matter over the course of the experiment. This decline was observed through field measurements with organic matter levels dropping from 32 g kg^{-1} to 25 g kg^{-1} of soil, probably as a result of starting the trial after a 5-6 year ley. Organic matter levels would be expected to rise again on return to a longer-term ley period, as is standard organic practice. The increase in organic matter levels from the implementation of a one-year fertility building ley was reflected in the NDICEA model, which showed a rise during this period, although the subsequent decline more than offset the gain. When a nutrient demanding crop was introduced into the rotation (e.g. potatoes in EFRC B) the deficit for all three nutrients increased to the extent that further use of external inputs (e.g. composts or manures) would be required, in particular for P and K. The high P and K offtake of beans similarly contributed to the large deficit of these two nutrients within the balance for rotation EFRC C.

Table 5.4: Nutrient balance of stockless organic rotations/trial sites expressed in kg ha^{-1} year^{-1}

	EFRC A			EFRC B			EFRC C			ADAS			HuntsMill Area 1			HuntsMill Area 6		
	N	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**
Fertiliser applied		8			8			8			8		8	5	18			
Deposition	20		4	20		4	20		4	30	1	5	20		3	20		3
Biological fixation	44			59			59			84			42			21		
Total supply	*64*	*8*	*4*	*79*	*8*	*4*	*79*	*8*	*4*	*114*	*9*	*5*	*70*	*5*	*22*	*41*	*0*	*3*
Volatilisation	0			0			0						0			0		
Denitrification	47			54			44			26			24			7		
Leaching	57			57			52			28			18			15		
Product Removal	36	8	6	63	17	32	55	14	26	77	15	45	35	9	55	33	10	62
Total loss	*140*	*8*	*6*	*174*	*17*	*32*	*151*	*14*	*26*	*131*	*15*	*45*	*77*	*9*	*55*	*55*	*10*	*62*
Nutrient balance	**-76**	**0**	**-2**	**-95**	**-9**	**-28**	**-72**	**-6**	**-22**	**-17**	**-6**	**-40**	**-7**	**-4**	**-33**	**-14**	**-10**	**-59**
Change in soil organic N	-93			-100			-60			0			-1			-13		
Change in soil mineral N	17			5			-12			-17			-6			-1		

A similar picture is presented for the rotation at ADAS Terrington. Despite the large contribution of N though fixation, there are considerable losses from leaching and denitrification, resulting in a negative N balance. It may be possible to address this problem through better use of over-winter cover crops (cover crops performed poorly over the course of the experiment with only one crop yielding over 1t DM ha^{-1}). The deficits for P within this trial are also unsustainable in the long-term and would need to be addressed through imports or by reducing the exploitative phase of the rotation. The K deficits for this site are also substantial, however it is possible that weathering of K stocks in the mineral pool could redress this (Khan *et al.*, 2014).

Lower rates of leaching were found for both Hunts Mill plots, although considerable deficits of P and K were also found despite the addition of green waste compost on area 1. The K deficits could be addressed through the buffering capacity of K-bearing minerals (Khan *et al.*, 2014) however the P deficit represents a more fundamental issue for the maintenance of crop yields in the longer term (Cordell *et al.*, 2009). The results for Hunts Mill area 6 also illustrate that it is possible to maintain a fairly balanced system with regard to N through the effective use of late summer/autumn sown green manures (i.e. without the use of a ley/break crop), although the overall deficit for N may result in a reduction in offtake or the need to use imported composts or manure.

Most of the stocked rotations were found to be more balanced with regard to N and P supply and loss (see Table 5.5). However, all of the SRUC sites faced a large K deficit, due to the high offtake from grass/clover silage, the potato crop (only at Woodside) and the use of straw for bedding, which were not offset by the manure application. Similar results were found within the nutrient balances for Tulloch and Woodside calculated by Watson *et al.* (2000) although lower offtake was estimated within this study due to the lower assumptions on P and K content within NDICEA. Despite the K deficit, no trend in K levels was found over time at Tulloch or Woodside, although the soil samples were restricted to the first 30cm and 45cm due to the presence of indurated layers at deeper levels, largely impenetrable to soil augers or crop roots. It is possible that soil K levels at these sites were being supplemented by reserves within parent material, in addition to potential inputs from crop residues (these inputs are ignored in standard soil K measurements and this in part explains why test values are unrelated to crop K balances, Khan *et al.*, 2014). These and other factors lead Khan *et al.* (2014) to suggest that measurements of available soil K are an unreliable indicator and that producers should use strip trials to determine site-specific fertiliser management. At both Woodside sites, the rate of N leaching was higher than at Tulloch, despite a lower annual rainfall. This was in part related to the lighter soil texture and the relatively low yield of the grass/clover leys at Woodside 37 (4-6 t DM ha^{-1}, Taylor *et al.*, 2006) which was related to the high soil moisture deficit. The low grass/clover yield at Woodside 37 also led to a negative N balance overall, due to a lower rate of biological N fixation. Much of the excess nitrogen at the other SRUC sites was locked up as organic matter (illustrated as a positive value under 'Change in organic N' in Table 5.5). This was observed within the trial through a small increase in SOM levels observed at Tulloch, although the measured organic matter levels at Woodside remained relatively constant. Volatilization rates were low across all of the stocked rotations in Table 5.5 as a result of incorporating applied manure on the same day as application on the trial sites.

Table 5.5: Nutrient balance of stocked organic rotations/trial sites expressed in kg ha^{-1} yr^{-1}

	Tulloch T50			Tulloch T67			Woodside W50			Woodside W37			Hunts Mill Area 1			Hunts Mill Area 6		
	N	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**	**N**	**P**	**K**
Fertiliser applied	63	17	63	83	22	81	63	17	62	38	10	37	48	13	53	62	16	61
Deposition	12	1	7	12	1	7	12	1	7	12	1	7	20		4	20		3
Biological fixation	57			109			112	0	0	60			25			9		
Total supply	*132*	*18*	*69*	*204*	*24*	*88*	*187*	*18*	*69*	*110*	*11*	*44*	*93*	*13*	*57*	*91*	*16*	*65*
Volatilisation	7			8			6			5			5			5		
Denitrification	3			3			12			11			15			14		
Leaching	26			30			49			45			29			29		
Product Removal	59	17	96	57	36	188	78	19	92	61	17	79	43	10	59	42	11	71
Total loss	*95*	*17*	*96*	*98*	*36*	*188*	*145*	*19*	*92*	*122*	*17*	*79*	*92*	*10*	*59*	*90*	*11*	*71*
Nutrient balance	**37**	**1**	**-27**	**106**	**-12**	**-100**	**42**	**0**	**-23**	**-12**	**-5**	**-35**	**1**	**4**	**-2**	**1**	**4**	**-6**
Change in organic N	32			101			41			6			1			1		
Change in mineral N	5			5			1			-18			0			0		

5.7.3 Nutrient (NPK) balance for typical rotations applied using site conditions of the organic trials

Nutrient balances are presented in Table 5.6 for each of the typical rotations described in Table 5.2. The results presented are mean values across all six sites and associated soil/weather conditions.

The stocked complex rotation described below seems to represent a well-balanced system with regard to N and P supply and offtake. However, the model predicted a relatively large K deficit with offtake exceeding supply. As discussed earlier, this could be addressed through K delivery from the weathering of minerals depending on the underlying geology, climatic conditions and site management (Khan *et al.*, 2014; Simonsson *et al.*, 2007), or through imported compost and/or mineral sources. The higher proportion of nutrient-demanding crops (e.g. potatoes) within the stocked simple rotation creates a larger K deficit compared to the stocked complex example. As with the stocked experimental sites in Table 5, volatilization rates were low for all of the stocked rotations in Table 6, as a result of selecting same-day incorporation of manure applied within NDICEA, and the low stocking density (i.e. 0.3 LU ha^{-1}). The volatilisation losses would be expected to increase if incorporation was delayed for any reason.

The stockless complex rotation in Table 5.6 has a deficit for all three nutrients. Two years of a red clover ley plus one year of spring beans did not provide enough N to support four years of crop offtake due to a high rate of leaching and denitrification. The presence of nutrient demanding crops contributes to the deficit (i.e. potatoes lead to a high N and K demand and beans to a high K offtake). The stockless simple rotation faces less of a deficit with respect to N, due to a higher input of biologically fixed N from the inclusion of peas which have a higher rate of N fixation than beans within NDICEA, and the use of the grass/vetch over-winter green manures following the spring crops. In addition there is an absence of nutrient demanding crops (e.g. potatoes) however the rate of leaching is still high.

Table 5.6: Nutrient balance of typical organic rotations/trial sites expressed in kg ha^{-1} yr^{-1}. se = standard error

	Stocked complex				**Stocked simple**				**Stockless complex**				**Stockless simple**			
	N	***se (+/-)***	**P**	**K**	**N**	***se (+/-)***	**P**	**K**	**N**	***se (+/-)***	**P**	**K**	**N**	***se (+/-)***	**P**	**K**
Fertiliser applied	32	*0*	12	23	26	*0*	14	21			8				9	
Deposition	17	*3*		5	17	*2*		5	17	*3*		5	16	*2*		5
Biological fixation	142	*9*			117	*9*			93	*10*			130	*11*		
Total supply	*190*	*9*	*12*	*28*	*160*	*10*	*14*	*26*	*109*	*10*	*8*	*5*	*146*	*11*	*9*	*5*
Volatilisation	2	*0*			2	*0*			0	*0*			0	*0*		
Denitrification	21	*2*			17	*2*			31	*3*			26	*2*		
Leaching	35	*9*			39	*9*			50	*10*			57	*11*		
Product Removal	79	*0*	13	37	72	*0*	14	42	49	*0*	10	37	55	*0*	10	16
Total loss	*137*	*8.7*	*13*	*37*	*130*	*10*	*14*	*42*	*130*	*10*	*10*	*37*	*138*	*11*	*10*	*16*
Nutrient balance	**53**	**7**	**-1**	**-9**	**30**	**9**	**0**	**-16**	**-21**	**3**	**-2**	**-32**	**8**	**3**	**-1**	**-11**
Change in organic N	51	*8*			28	*8*			-3	*10*			20	*9*		
Change in mineral N	2	*8*			2	*7*			-18	*9*			-12	7		

The relatively low deficit of P within all of the typical rotations in Table 5.6 is a result of the application of rock phosphate. All of the modelled rotations would face a P deficit on a similar scale to the K balance without the use of this input.

5.7.4 Implications for improved organic management

In common with previous studies, the work presented here found considerable rates of N leaching within the rotations assessed (Berry *et al.*, 2003; Kirchmann *et al.*, 2007; Torstensson *et al.*, 2006). In some cases, this exceeded the amount lost by product removal (e.g. the stockless simple rotation described in Table 5.4). High rates of leaching under organic management are related to difficulties associated with matching crop N demand with N availability, particularly following incorporation of the ley, when N availability exceeds demand (Aronsson *et al.*, 2007; Bergström *et al.*; Torstensson *et al.*, 2006). The use of organic manures can also make it difficult to predict N availability, compared with applications of mineral fertiliser (Cassman *et al.*, 2002), making it more difficult to maximise N recovery and crop yields under organic management (Seufert *et al.*, 2012). As a result of these factors, lower nitrogen use efficiency has been reported for organic cropping in comparisons with conventional systems (Bergström *et al.*, 2008; Torstensson *et al.*, 2006).

The effective use of over winter green manures and undersowing of leys in cereal crops will help to reduce losses and thus enhance overall N efficiency (Kaffka and Koepf, 1989; Torstensson *et al.*, 2006) and the lowest rates of leaching within this study were found for the rotations incorporating undersown crops and cover-crops (e.g. ADAS Terrington, Warwick University Hunts Mill). Poor cover-crop establishment (e.g. at the ADAS and Elm Farm experiment) was experienced as a result of competition from weeds and slow emergence, which reduced the benefit obtained (ADAS, 2006). Poor cover-crop establishment can also be related to competition from the cash crop, adverse weather conditions and low soil temperatures at the time of sowing (Snapp *et al.*, 2005). In particular, the occasional occurrence of poorly performing cover crops presents an important challenge for the long-term sustainability of stockless systems, which rely on keeping the N supplied through biological fixation within the system. Although with careful rotation design, such systems are, in theory, sustainable from a nitrogen management perspective (Schmutz *et al.*, 2007), in practice these systems appear to be highly vulnerable to poor establishment during the cover-cropping period.

The use of cover-crops is not limited to organic farms, and higher nitrogen use efficiencies can be obtained by using this method alongside targeted mineral fertiliser application(s) to meet crop demand (and thus increase yield) whilst minimising losses (Torstensson *et al.*, 2006). Such tightly controlled systems could represent a suitable approach to developing highly N-efficient production systems, through a combination of organic practices and targeted fertiliser application (Cassman *et al.* 2002, Godfray, 2014). Similar targeted approaches could still be used on organically-managed land, through the use organic fertilisers with a high N availability (e.g. poultry manure and digestate from slurry based anaerobic digestion) to supply readily available N at key points in the rotation (Berry *et al.*, 2002; Möller and Stinner, 2009). However the application of such sources can increase the occurrence of nitrophilous weeds and their use within organic systems has been questioned as the high N availability leads to feeding the plant instead of the soil (Möller, 2009) and a reduction in the amount of organic matter applied in the case of digestate (Oelofse *et al.*, 2013; Stinner *et al.*, 2008). The use of perennial crops can also help to reduce leaching in organic systems through keeping the soil covered and improving N synchrony (Cox *et al.*, 2002; Di and Cameron, 2002), although lower yields, weed susceptibility and pest and disease management issues may limit uptake (Pimentel *et al.*, 2012). A lack of technical information, suitable varieties and socioeconomic constraints (e.g. lower consumer demand compared to staple annual crops) also limit the potential for a wider adoption of perennial cropping (Pimentel *et al.*, 2012; Valdivia *et al.*, 2012).

Organic farmers can also reduce N leaching considerably through improved management of manures and slurries. In particular careful storage, application timing and choice of application method will help to maximise N recovery and minimise losses where slurries and manures are applied (Smith *et al.*, 2002; Webb *et al.*, 2010). Manure analysis can also improve on farm nutrient use efficiency and help to reduce losses by improving understanding of nutrient supply from organic sources (Watson *et al.*, 2005). In some regions, there may be opportunities for farmers to work together to measure the nutrient use efficiency of their systems through a combination of manure and livestock dietary analysis combined with soil sampling (Le Gal *et al.*, 2011; Verhoeven *et al.*, 2003). Such participatory approaches can be effective at allowing for improvement options to be identified and for the fine-tuning of production systems. Again the use of such methods is not restricted to organic farms, however the inability of such farms to access manufactured N fertiliser makes the implementation of such measures all the more important for the effective prediction of N supply.

With regard to phosphorus, the modelled systems able to achieve a sustainable balance were using external inputs of rock phosphate to offset losses. Although rock phosphate can help to offset losses, a reliance on this source may result in limited P bioavailability to meet crop demand, due to slow rates of solubilisation (Edwards *et al.*, 2010). In addition, the use of such a fertiliser clearly does not fit well with the International Federation of Organic Agriculture Movements (IFOAM) organic principles (IFOAM, 2006) which emphasise the importance of reducing inputs to increase the long-term sustainability of farming systems. Despite this aim, the use of imported manure, straw and/or rock phosphate is common on organic farms, particularly for the supply of P and K (Kirchmann *et al.*, 2008; Nowak *et al.*, 2013; Oelofse *et al.*, 2010). In many cases, manure and straw is sourced from conventional farms, which has led to the conclusion that organic farms are being 'propped-up' by conventional agriculture, and that as a result a large-scale conversion to organic management would be unsustainable (Kirchmann *et al.*, 2008; Nowak *et al.*, 2013; Oelofse *et al.*, 2010). Organic monogastric systems (in particular poultry) also often require imported feed (e.g. soy) to supply protein and essential amino acids (Dekker *et al.*, 2011) and so these systems are supplemented by internationally imported P and K.

The use of household waste and sewage sludge on organic farms could represent a possible solution to reduce the reliance on conventional manure and/or rock phosphate on organic cropping farms, in particular for the sustainable supply of P. Source separated urine also presents an opportunity to apply readily available nitrogen and phosphorus (Germer *et al.*, 2011; Karak and Bhattacharyya, 2011; Kirchmann *et al.*, 2008). The use of such sources clearly fits with the organic ideal of closing the system as far as possible (Oelofse *et al.*, 2013), although in this case the 'system' expands beyond the farm gate to the consumer (Oelofse *et al.*, 2010). Although there have been many cases of household waste recycling on organic farms to supply nutrients to the soil (Altieri *et al.*, 1999; Darnhofer, 2005; Luske and van der Kamp, 2009), the use of sewage sludge or urine is strictly prohibited on organic land in Europe, despite the fact that its use seems to be a rational and scientifically supported method of closing the nutrient cycle. Developments in the area of struvite (magnesium ammonium phosphate) recovery from waste water treatment plants could present a possible solution, allowing for application of a refined and slow release mineral fertiliser product, however this product is not currently on the list of permitted fertiliser within the European Commission organic regulation (European Council, 2007). This is an area that needs further scrutiny from a scientifically based perspective as it would appear that historical concerns about the toxic effects of applying urine and sewage sludge to agricultural land may no longer be justified (Karak and Bhattacharyya, 2011; Smith, 2009), although public perception concerning the risks to human health remains an issue in some areas (Robinson *et al.*, 2012). Increasing the co-operative use of manure between (organic) livestock and arable farmers has also been suggested as a possible route for reducing the use of conventional manures on organic land and, within farming in general, the co-operative use of manure between specialised livestock and arable holdings could contribute to the prevention of stockpiling of nutrients and associated losses on intensive livestock holdings (El-Hage Scialabba and Müller-Lindenlauf, 2010a; Wilkins, 2008). In particular this approach has been encouraged in Denmark by a decision to phase out the use of conventional manure and straw on organic land by 2021, partly in recognition of the conflict between principle and practice and partly to prevent the import of genetically modified organisms (GMOs) into organic systems via manure (Oelofse *et al.*, 2013). In addition, the transition strategy in Denmark has highlighted the importance of crop rotation design (in particular to improve understanding on nutrient supply and losses), the development of crop cultivars for low-nutrient environments, and the development of biogas plants that can run on plant-based feedstock (in particular grass/clover harvested from leys) in recognition of the limited supply of organic manure (Jørgensen and Kristensen, 2010; Oelofse *et al.*, 2013).

Potassium deficits were observed across all of the rotations however on many soils, this does not present an issue given vast reserves of mineral K within parent material which may be released for plant uptake by weathering (Khan *et al.*, 2014). Despite this potential, Holmqvist *et al.* (2003) found that weathering and bioavailability from the mineral fraction can vary greatly (between 3 and 80 kg K ha^{-1} yr^{-1} on a range of soil types in Norway, Sweden and Scotland) although the modelled predictions in this study did not take into account the dynamic and localised biological weathering by plant roots illustrated by x-ray diffraction studies (e.g. Hinsinger *et al.* (1991) in Khan *et al.* (2014)) and the potential contribution of mycorrhizal fungae to K availability (Hoffland *et al.*, 2002). Nevertheless improved knowledge of site-specific geochemical and mineralogical data in addition to soil rhizosphere interactions, could be a useful aid to the development of site-specific fertiliser recommendations and nutrient balances (Andrist-Rangel *et al.*, 2007; Holmqvist *et al.*, 2003). With respect to mineral reserves of K on the sites assessed in this study, only EFRC, IBERS and ADAS Terrington could be expected to supply a considerable amount of K from the clay fraction (Buckman and Brady, 1984), although sand- and silt-sized muscovite and biotite can also be a major source of plant-available K on lighter soils (Mengel *et al.*, 1998) and the presence of these and other K-bearing minerals may have offset some or all of the K offtake at Hunts Mill and the Scottish sites (Andrist-Rangel *et al.*, 2010). Despite the high deficits, there was no apparent trend in available K levels over time at most of the experimental sites considered, although Hunts Mill showed a slight decline over the course of the study and the K measurements at Tulloch (taken in the winter) may have been affected by the preceding silage crop (Watson *et al.*, 2000). Other studies have demonstrated a decline in soil P and K levels following conversion to organic management (Loes and Øgaard, 1997; Torstensson *et al.*, 2006) and positive yield responses have been observed following K applications in long-term experiments in Australia and the UK (Bar-Yosef *et al.*, 2015) and within rice production systems, following several years of intensive cropping (Greenland, 1997). It is thus important to use nutrient budgets together with soil analysis to help understand the buffering capacity of soils and the management of P and K on individual fields. It should also be remembered that the bank-balance (i.e. supply minus offtake) concept of nutrient management can have major limitations, as N fertilisation in excess of crop removal can lead to a depletion of soil carbon reserves by enhancing microbial decomposition (Khan *et al.*, 2007; Mulvaney *et al.*, 2009). This approach can also lead to an uneconomical fertiliser usage in the case of K that may also have an adverse effects on soil quality and productivity (Khan *et al.*, 2014) although a range of management factors (e.g. N supply and tillage system) can mask the effect of K fertilisation on crop yield (Bar-Yosef *et al.*, 2015). It has also been suggested that crop yield and quality reductions following K fertiliser application are more likely to be related to K-Mg and K-Ca antagonism in plant uptake and/or K immobilisation in the soil (Bar-Yosef *et al.*, 2015), rather than toxicity in the plant and root zone, or a depletion of the soil structure (Khan *et al.*, 2014).

In summary, it is clear from the analysis and modelling within this study that most typical organic cropping systems in the UK will require nutrient inputs to maintain an N, P and K balance. It should also be remembered that most organic farms import fewer nutrients than their conventional counterparts (Hansen *et al.*, 2000; Torstensson *et al.*, 2006; Trydeman Knudsen *et al.*, 2006). Although this approach naturally leads to lower yields, and can lead to lower nitrogen efficiencies within cropping systems (Torstensson *et al.*, 2006), it can also offer a useful way to balance production and environmental concerns (Francis and Porter, 2011; Gomiero *et al.*, 2011). For example organic farms often require less fossil energy on a per hectare or kilogram of product basis, in particular through the absence of imported mineral N fertiliser (Smith *et al.*, 2015). The use of grass/clover leys and manures for fertility building on organic farms also contributes to greater SOC concentrations and stocks on organically managed land (Gattinger *et al.*, 2012). In addition organic methods (e.g. use of clover and other legumes to supply N) can be used effectively on conventional farms to increase efficiencies and reduce the environmental impacts of the agriculture sector as a whole (Gaudin *et al.*, 2013; Godfray, 2014; Pretty *et al.*, 2005).

5.8 Conclusion

An assessment of the NDICEA model has found that it is a useful tool for UK organic farmers to assess the amount of nitrogen supplied and lost through their rotations, although the model should be calibrated to improve accuracy for UK conditions where measured crop N, P, K, soil-N and organic matter values are available. The modelling of the N, P and K balance within organic trials found that in most cases sufficient N is being supplied through biological fixation to support the cropping, although leaching in higher rainfall areas and on lighter soil types may prevent the N from becoming available to the crop(s). The study has also shown that careful rotation design is particularly important within stockless organic systems to reduce losses and avoid the requirements for external inputs as far as possible. Although adequate nitrogen balances are theoretically achievable within stockless organic cropping systems, these systems are highly vulnerable to cover crop failure, poor crop yields and low rates of N fixation within the fertility building period. Negative P and K balances were found for most of the experimental stockless systems and the typical stockless rotations modelled within this study. For phosphorus, the systems seem to be dependent on imported rock phosphate for the maintenance of a small surplus or deficit. The much larger K deficits could be addressed through weathering and subsequent bioavailability of mineral K stocks, depending on site and management conditions. On soils with naturally low K deposits within parent material, K inputs in the form of fertiliser or feed may be required to offset removal, or a reduction in K demanding crops (e.g. potatoes) may be necessary.

N, P and K balances on organic farms are a useful method for exploring the extent to which organic methods can be applied effectively to improve nutrient use efficiencies within agricultural systems. It is likely that the greatest nitrogen use efficiencies can be achieved through a combination of organic production methods (e.g. use of cover crops and clover to supply N) combined with conventional farming practices (e.g. use of mineral fertiliser at key points in the rotation to meet crop demands fully and increase yields). In addition, the need to obtain minerals from sustainable sources leads to the conclusion that deriving these from suitably defined wastewater treatment could close the nutrient loop for organic farms, but this would require a change in international standards.

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CHAPTER 6. THE PRODUCTIONS IMPACTS OF A 100% CONVERSION TO ORGANIC FARMING IN ENGLAND AND WALES

Article title	Modelling the production impacts of a widespread conversion to organic agriculture in England and Wales
Journal and publication status	*Land Use Policy.* Submitted
Co – authors	Philip Jones (Reading University) Adrian Williams, Guy Kirk (Cranfield University), Bruce Pearce (The Organic Research Centre)
Co-author contributions	Assistance with model development, selection of scenarios and comments on draft(s) of written text
Research methods applied	Geographic Information System-based data extraction and processing National farm-level data extraction and analysis Linear programming in GAMS Sensitivity analysis

6.1 Abstract

We assess the production impacts of a 100% conversion to organic agriculture in England and Wales using a large-scale linear programming model. The model includes a range of typical farm structures, scaled up across the available land area, with the objective of maximising food production. The effects of soil and rainfall, nitrogen (N) supply/offtake and livestock feed demand are accounted for. Results reveal major reductions in wheat and barley production, whilst the production of minor cereals such as oats and rye increased. Monogastric livestock and milk production also decreased considerably, whilst beef and sheep numbers increased. Vegetable production was generally comparable to that under conventional farming. Minimising the area of fertility building leys and/or improving rates of N fixation increased the food supply from organic agriculture at the national level. The total food output, in terms of metabolisable energy, was 64% of that under conventional farming. This would necessitate substantial increases in food imports, with a corresponding expansion of cultivated agricultural land overseas. Significant changes in diet and reductions in food waste would be required to offset the production impacts of a 100% conversion to organic farming.

6.2 Introduction

The continuing expansion and intensification of global agriculture presents a clear need to develop modes of production that can supply sufficient amounts of food for growing populations with more efficient use of resources (Godfray *et al.*, 2010). At the same time there is a pressing need to move populations of western countries towards more balanced diets to promote public health, with particular regard to fresh vegetable and fruit consumption (Macdiarmid *et al.*, 2011; Wellesley *et al.*, 2015). Organic farming has the potential to contribute to developments in the first of these areas through a focus on reduced input intensity and the maintenance or enhancement of ecosystem functions and various studies have identified and quantified the benefits of organic production, in areas such as fossil-energy use, biodiversity, human nutrition and on-farm employment (Lampkin *et al.*, 2015). The significantly higher soil carbon sequestration rates observed in organically managed soils have also led to suggestions that wider use of this production system could help to delay the onset of damaging climate change (Gattinger *et al.*, 2012) although others note that the requirement to increase the area of land in agricultural production to meet food demand could offset these benefits (Leifeld *et al.*, 2013). The benefits provided by organic agriculture in areas such as soil protection and rural development also align with the dimensions of sustainability proposed by the United Nations following Rio+20 and EU action plans such as the Biodiversity Strategy (European Commission, 2010) and the Soil Thematic Strategy (European Commission, 2006).

While acknowledging these sustainability benefits and the potential for further growth in the market for organic products (Willer and Lernoud, 2016) some commentators (for example Connor, 2008) have suggested that the lower yields observed in organic agriculture would mean that widespread conversion to organic production could be detrimental to food security. Because the land area devoted to organic farming globally currently remains very small (i.e. organic farmland constitutes approximately 1% of the total global agricultural area, Willer and Lernoud, 2016), it is also difficult to extrapolate from this low baseline to assess the impacts of much larger scale adoption.

Despite this limitation, a few studies have attempted to explore the production and food security impacts of a widespread conversion to organic farming, the most recent of which was completed in 2009 by the Centre for Agricultural Strategy at the University of Reading, UK (Jones and Crane, 2009). In this study two different approaches were used to estimate how much food might be produced under an assumed 100% organic conversion of agriculture in England and Wales. The results indicated that full organic conversion would lead to major reductions in wheat, barley, and oilseed rape production. Pig and poultry numbers would also fall markedly, while there would be significant increases in the production of minor-cereals (e.g. oats, rye) and ruminant livestock. Although the Jones and Crane (2009) study projected credible trends, levels of production were not adjusted in line with N availability (i.e. the nitrogen availability constraints that impact organic farming, Berry *et al.*, 2002). Feed availability and the nutritional requirements of livestock were also not assessed in detail. Prior to this 2009 study, Badgley *et al.* (2007) assessed the implications of a 100% conversion to organic production at a global level using FAO-derived data. Organic yield adjustment coefficients (i.e. organic versus conventional) were estimated for 10 groups of crops and livestock products, based on a review of 293 studies drawn from the peer-reviewed literature. At the global level Badgley *et al.* (2007) estimated the average organic yield ratio for all crop types to be 1.32 (i.e. organic would produce 132% of the conventional yield). The total N supplied by leguminous cover crops in organic systems was estimated to be 140 million Mg which, according to the study authors "is 58 million Mg greater than the amount of synthetic N currently in use" (Badgley *et al.*, 2007). The authors therefore suggest that the rates of biologically fixed N under widespread organic conversion could support yields equivalent to high-yielding conventional agriculture. Although the Badgley *et al.* (2007) study included estimates of N availability, the authors base this on the erroneous assumption that 100% of arable land could accept an additional legume crop, following the main crop in the same year. In making this assumption the authors fail to account for the fact that much of the world's productive land area is already required to carry multiple food crops in a single year to meet food demand. Additionally, no account was taken of areas where climatic conditions and water supplies limit the possibility of a second crop in the same year (Connor, 2008).

The study presented here builds on these earlier studies to provide a robust estimate of the production and food security impacts of a 100% conversion to organic farming in England and Wales. A modelling approach was adopted that was able to account for yield differences between conventional and organic production, as well as yield variation due to local environmental conditions, plus supply constraints imposed by the availability of N, the need to maintain rational crop rotations, and the availability of livestock feeds. A multi-scenario approach was adopted to explore the impact of variation in the assumptions underpinning these constraints. In addition, a healthy eating framework developed in the UK was used to assess the ability of a fully organic domestic agriculture to supply optimal human nutritional requirements (i.e. the Eatwell Plate, Macdiarmid *et al.*, 2011).

6.3 Methods

6.3.1 The OLUM model

A linear programming model was developed – the Optimal Land Use Model (OLUM) – in the GAMS programming language (GAMS Development Corporation, http://www.gams.com/), to explore the impacts of 100% conversion to organic farming in England and Wales. Figure 6.1 summarises the model. At its core is an objective function, Z, to maximise the output of food (expressed as metabolisable energy - ME), defined as:

$$\mathbf{Z} = \sum_{ij=0}^{n} \boldsymbol{C_{ij}} \cdot \boldsymbol{x_{ij}} \quad \text{subject to} \quad \boldsymbol{Rx_{ij}} \leq \mathbf{b},\ \boldsymbol{x_{ij}} \geq \mathbf{0} \tag{1}$$

where C_{ij} is the ME output per unit of agricultural products *i* on soil × rainfall class j and x_{ij} is a scalar, i.e. areas of crops and numbers of livestock produced. Rx_{ij} is the resource (R) requirement of producing enterprises (x_{ij}) and b is the resource endowment and input availability vector. Constraints are specified as linear inequalities and equalities and employed to determine the following:

1. Availability of land by farm type and soil × rainfall class.
2. Maximum and minimum stocking densities (livestock units per ha).
3. Annual feed requirements of different livestock and feed types, expressed as metabolisable energy (ME) and crude protein (CP) requirements.
4. Maximum/minimum crop areas by crop groups (i.e. rotation constraints).

5. Soil N availability reflecting cycling of nutrients, plus N inputs and outputs through crop and livestock offtake, atmospheric deposition and biological N fixation.
6. Upper limits on the total permissible production volumes of individual crop and livestock products, set at 150% of the current supply, on the assumption that increases beyond this volume could not be absorbed by the market. Geographical constraints on sugar-beet production were also imposed to restrict the expansion of this crop away from major processing centres in eastern regions.

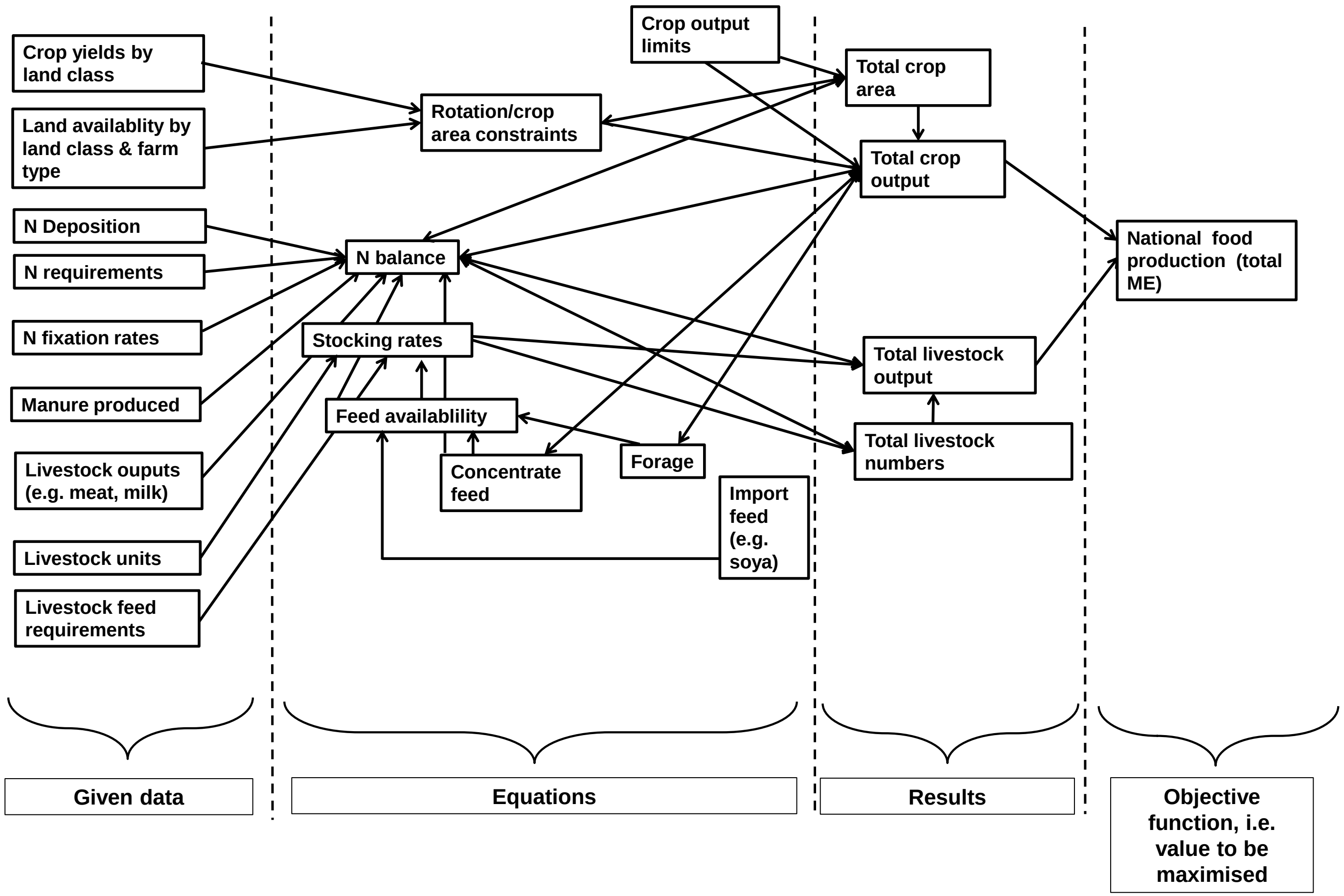

Figure 6.1. Schematic of the Optimal Land Use Model (OLUM).

The components of the model are as follows.

6.3.2 Farm types

The functional units are farms, i.e. systems consuming various inputs, including land and other resources, to produce multiple crop and livestock outputs. Nine farm types are defined based on the Defra Robust Farm Types (Figure 6.2). The mix of enterprises available to each farm type was fixed, although the model was permitted to vary the relative scale of these. This constraint was based on the observation that the dominant enterprises on farms under conventional agriculture is usually maintained post-conversion, because these are the activities that suit existing farm infrastructure and local conditions (Howlett *et al.*, 2002; Langer, 2002).

6.3.3 The land base

Land availability was fixed, at the national level, within NUTS1 region and within farm type. Within each farm type, the allocated land area was fixed at the area under each Robust Farm Type observed in the 2010 Defra June Survey of Agriculture. It was assumed that the total land area under each robust farm type would not change following organic conversion. The land base was disaggregated into 16 classes based on soil type and rainfall (next section) and yield potential was determined for each class. Within each farm type and NUTS1 region, the areas of these 16 land classes were fixed according to their observed spatial distribution.

6.3.4 Land classes

Heavy, medium and light soil classes were specified, each with estimated organic matter content and pH values based on data from long-term organic cropping trials (Smith *et al.*, 2016). A fourth soil class was specified for 'humose', i.e. cultivated soils with an organic matter content and pH typical of the Downholland soil series of the Soil Survey of England and Wales (www.LandIS.org.uk), which is representative of such soils. The spatial distribution of each soil class in 5 km × 5 km grid squares across England and Wales was obtained from the National Soil Inventory (www.LandIS.org.uk). Four rainfall classes were specified based on 30-year Meteorological Office annual rainfall data. These were, dry 539–635 mm, medium 636–723 mm, wet 724–823 mm and very wet >824 mm. To determine the total areas of each soil × rainfall combination (hereinafter 'land classes'), the dominant combination was identified in each of the 5 km × 5 km grid squares of the National Soil Inventory, and then the sum of the areas of each square, less any non-agricultural area, was allocated to that land class (Figure 6.3).

The areas of each land class within each farm type and NUTS1 region were estimated to generate constraints on land availability at these levels. The sum of these areas provided the constraint on land availability at the national level:

$$\sum_{c=0}^{n} a_{c,t,s,r} = L_{t,s,r} \quad \forall \ t,s,r \tag{2}$$

where $a_{c,t,s,r}$ is total production area, summed over for each crop (c), farm type (t), land class (s) and NUTS1 region (r) and $L_{t,s,r}$ is total land availability.

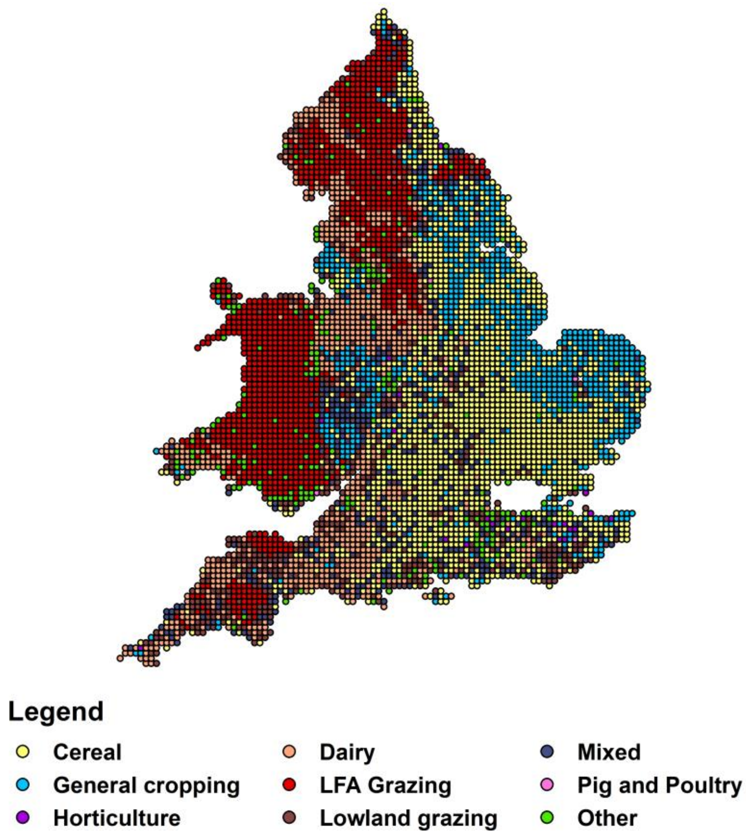

Figure 6.2. Dominant Robust Farm Types on a 5 km × 5 km grid across England and Wales. Data are from the Defra June Agricultural Census (Defra, 2011).

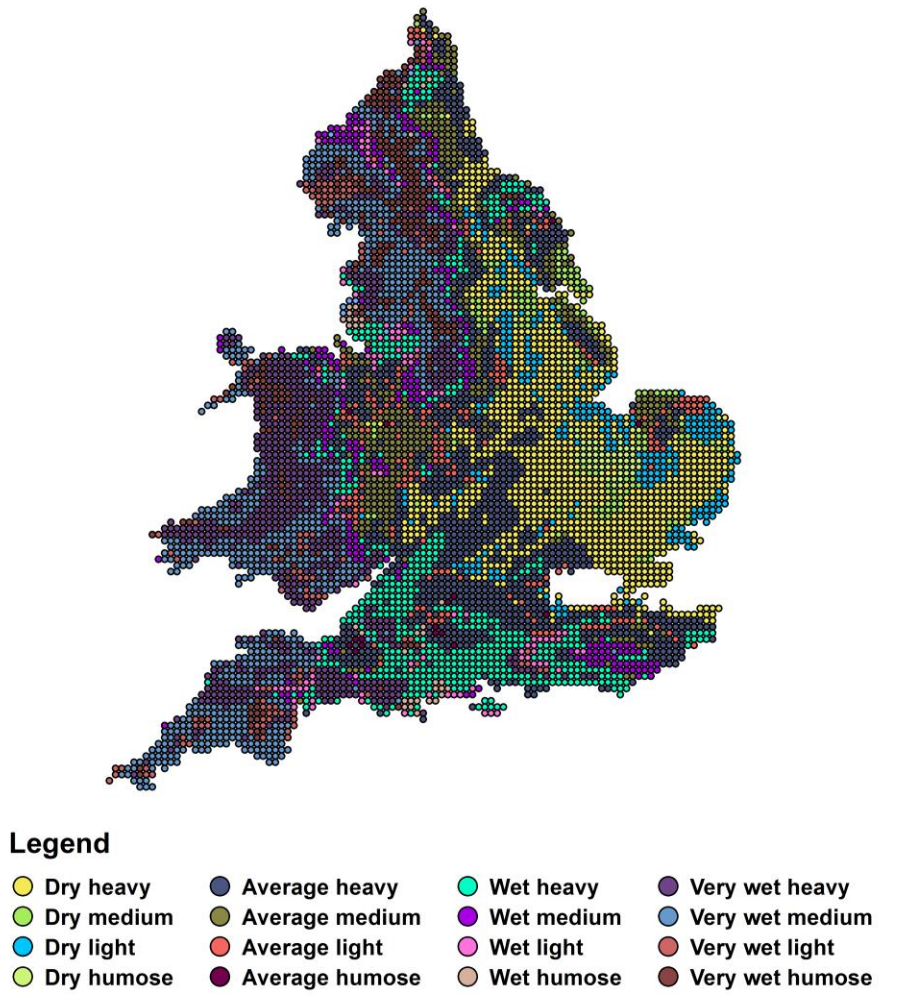

Figure 6.3. Dominant land classes based on soil type and rainfall on a 5 km × 5 km grid . Data sources are described in the main text.

6.3.5 Crop yields

Potential crop yields for each land class were estimated using the Nitrogen Dynamics in Crop Rotations in Ecological Agriculture (NDICEA) model (van der Burgt *et al.*, 2006). Smith *et al.* (2016) showed that NDICEA gives sufficiently accurate estimates of N availability for our purposes in a range of UK soil types and rainfall zones, using data from long-term organic trials. NDICEA has three modules as follows:

- *soil water dynamics*, which accounts for irrigation, rainfall, evapo-transpiration, capillary rise and percolation;

- *N mineralisation*, which accounts for N availability from soil organic matter (SOM) and organic manure; and
- *inorganic N dynamics*, which accounts for N inputs from mineralisation, atmospheric deposition, fertilisers, irrigation and biological fixation, and N losses through denitrification, leaching and crop uptake.

NDICEA is target-oriented, meaning target yields are entered by the user and adjusted manually. We thus iteratively adjusted yields according to N supply in each land class for the rotations in Table 6.1. Points in the rotation where N availability was greater or less than crop requirements were identified, and yields were adjusted accordingly up to a maximum yield potential, based on data sources described in Appendix B.

Table 6.1. Rotations assessed within NDICEA to derive crop yields for each soil and rainfall class

	Rotation year									
Rotation	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**
Stocked 'complex'	G/WC	G/WC	G/WC	WW	WO	RC/G	RC/G	P	SB	SW
Stocked 'simple'	RC/G	RC/G	WW	P	WW	WR				
Stockless 'complex'	RC/G	RC/G	P	WO	SB	SW				
Stockless 'simple'	RC/G	WW	PE	SO						
Field vegetable	RC/G	RC/G	P	BR	L					
Market garden	RC/G	RC/G	CB	O	B	C	SB	BR	PE	CG
Dairy	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	FB	WS	SB
Cattle and sheep	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	G/WC	FB	WW
Mixed	G/WC	G/WC	G/WC	RC/G	WW	WO	SB	WB	WR	

(G/WC = Grass/white clover, WS = wholecrop silage, WB = winter barley, WW = winter wheat, WO = Winter oats, RC/G = red clover, SW = spring wheat, SB = Spring beans, P = potatoes, WR = Winter rye, FB = Fodder beet, PE = peas, SO = spring oats, BR = broccoli, L = leeks, CB = cabbage, O = onions, B = beetroot, C = carrots, CG = courgettes, SB = spring barley)

Example results from the NDICEA yield estimation and adjustment exercise are shown in Appendix B. Due to the lack of yield data for organic oilseed rape and sugar beet in the UK, as a consequence of very limited production of these organic crops, yield data from a national survey of organic farmers in France and a UK-based modelling study were used (Tzilivakis *et al.*, 2005; Valantin-Morison and Meynard, 2008). The yields for these crops were adjusted for each of the 16 soil/rainfall classes on the same basis as crops considered similar in terms of their likely position in the rotation (i.e. wheat and potatoes).

6.3.6 Grass yields

A regression-equation model, based on the grass site class system of Brockman and Gwynn (1988), was used to estimate organic permanent pasture yields based on annual rainfall, soil type and altitude (Williams *et al.*, 2006). The model was validated by comparison with yield data from grassland-dominated organic conversion trials at the University of Wales, Aberystwyth (Haggar and Padel, 1996) and Scotland's Rural College (SRUC,Taylor *et al.*, 2006). Appendix B describes the regression model and the calculated yields.

6.3.7 Crop rotation constraints

Crop rotation is a necessary component of organic systems to break pest and disease cycles, control weeds and maintain soil N through biological fixation (Lampkin, 2002). It was therefore important to include rotational constraints in the OLUM on the area of each crop type that could be grown. The rotational constraints were applied at the level of crop group defined in terms of common growth characteristics, i.e. similar nutrient requirements and pest/disease susceptibility (see Appendix B). The minima and maxima area constraints on these crop groups were derived from the rotational data described in Table 1 and specified in the model by:

$$\sum_{g=0}^{n} a_{g,t,s,r} \;\geq \text{or} \leq\; L_{t,s,r} \cdot R_{g,t} \qquad \forall\; t,s,r \tag{3}$$

Where a is the total land-area produced by crop group (*g*), in each farm type, soil/rainfall class and region (t,s,r) , $L_{t,s,r}$ is total land availability by farm type, land class and region and $R_{g,t}$ is a coefficient reflecting the minimum or maximum proportion of total utilisable agricultural area (UAA) that must be allocated to this crop group.

6.3.8 Constraints on livestock numbers

Total livestock numbers were constrained both within farm type and at the national level. At the farm-type level, permissible maximum and minimum stocking rates were set, reflecting constraints inherent in CAP cross-compliance measures. These stocking rates were derived from actual practice, as observed in the organic sub-sample of the Defra Farm Business Survey (Moakes *et al.* 2012, 2014). Minimum and maximum stocking rates averaged over the organic farm sample over a three year period were calculated by dividing total livestock units by total land area. At the national level maximum permissible livestock units of each stock type were set, and a separate constraint was set through a maximum manure-N production of 170 kg-N per hectare averaged over the entire land base (a limit set for organic production within Council Regulation No 889/2008, 2008). As the data provided by Moakes et al (2012,2014) excludes information on organic poultry and pig farms, alternative sources were used (i.e. Browning pers. comm. 2016, Leinonen *et al.*, 2012a; Leinonen *et al.*, 2012b) to derive stocking rate limits for these livestock types.

Minimum and maximum stocking rates were defined by:

$$\sum_{l=1}^{n} lu_l \cdot l_{l,t,s,r} \geq \text{or} \leq \sum_{c=1}^{n} c_{c,t,s,r} \cdot sr_t \quad \forall \; t,s,r \tag{4}$$

where lu_l is a standard livestock unit conversion factor, $l_{l,t,s,r}$ is livestock numbers, $c_{c,t,s,r}$ the total agricultural area and sr_t the minimum or maximum stocking rate per ha within each farm type, soil/rain class and region (t,s,r). Minimum stocking rate constraints were removed for specialist cereals, field vegetables, market gardens and general cropping farms to allow for stockless production.

The numbers of young stock, replacements and other stock (e.g. pigs and poultry) required by the model were calculated as a fixed ratio of the numbers of adult animals in the dominant livestock type on each farm type (i.e. the stock type with the highest number of livestock units as a proportion of the total livestock presence). An example of the approach used is:

$$\sum_{bc=1}^{n} \sum_{lg=1}^{n} \sum_{s=1}^{n} \sum_{r=1}^{n} l_{bc,lg,s,r} \cdot pl_{fc} = l_{fc,s,r} \quad \forall \; l,lg,s,r \tag{5}$$

where $l_{bc,lg,s,r}$ is the number of adult animals of the dominant livestock type (in this example beef suckler cattle bc) pl_{fc} is a fixed proportion reflecting the number of replacements required to maintain the adult herd. The term $l_{fc,s,r}$represents total store cattle numbers and lg is the area of the farm-type "lowland-grazing" in each soil/rain class (s) and region (r).

6.3.9 Feed availability

Livestock numbers were also limited by total feed demand and availability. The ME requirements of the livestock produced were offset against the ME availability in the feedstocks produced. Data on the ME requirements of the different types of livestock and the energy and protein contents of different types of crops and grasses, plus purchased feeds, were drawn from a range of industry sources and technical guides (Lampkin *et al.*, 2014; Soffe, 2003; The Professional Nutrient Management Group, 2015). Livestock concentrate feed composition data were obtained from Vitrition Organic Feeds, Newcastle University (Edwards, 2002) and a recent study on the feasibility of replacing soy in UK livestock production with UK-grown protein crops (Jones *et al.*, 2014). Feed supply constraints and minimum feeding requirements for different types of livestock were defined using the following feed-groups:

- forage (e.g. grass/clover, fodder beet, fodder maize);
- concentrates/straights (e.g. cereals, beans, peas); and
- compound feeds (processed feeds incorporating straights, plus other supplements including, soybean and oilseed meals, crop processing residues, and other imported feed including molasses).

The proportion of the total livestock ME requirement supplied by each feed group was predetermined for each robust farm type, using data reported in Moakes *et al.* (2012, 2014). Due to the dominance of forage crops within most organic rotations (e.g. grass, clover and other leguminous crops) a maximum forage ME supply was applied at the farm type level in each region to reflect the fact that most organic farms still feed some concentrate and compound feed for finishing. This constraint also ensured that the ME from forages demanded by ruminant livestock did not exceed the ME available from the forage crops. This constraint was applied within each farm type and region to reflect the fact that forage is unlikely to be transported between farms due to costs and impracticalities associated with the transport of such high bulk, low value products. More details on the livestock feed constraints applied are contained in the detailed model description (Appendix C).

6.3.10 Livestock outputs

Outputs of beef, sheep meat and milk were constrained, to reflect current yield potential, on a livestock headage basis for each farm type and region using financial output data from Moakes *et al.* (2013), financial and physical output data from the Organic Farm Management Handbook (i.e. total financial output by livestock type divided by price/head, Lampkin *et al.*, 2011) and a recent study by the Agriculture and Horticulture Development Board (AHDB Dairy, 2012). Eggs, poultry meat and pork production were also constrained per head of livestock based on rearing periods and liveweights recorded by Leinonen *et al.* (2012a, b) and Soffe (2003). These data were applied through equations which expressed total volume/value of output as a proportion of the headage of livestock on each farm (more on this approach is provided in Appendix C).

6.3.11 Nitrogen balances of crops and livestock

As the supply of N can be a limiting factor for the maintenance of productivity in organic systems (Berry *et al.*, 2002) N supply and offtake equations were incorporated within OLUM. Total N supply and crop/livestock offtake were defined on a regional basis to allow for transfer of manure between farms within the same area, as in Equation 6:

$$\sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} co_{c,t,s,r} \cdot a_{ch,t,s,r} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} livn_{l} \cdot l_{l,t,s,r} \leq$$

$$\sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} fx_{c,t,s,r} \cdot a_{c,t,s,r} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} lu_{l,t,s,r} \cdot l_{l,t,s,r} \cdot Nin_{t} +$$

$$\sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} imp_{ltsr} \cdot compn + \sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} a_{c,t,s,r} \cdot dp_{c,t,s,r} \quad \forall\, r \qquad (6)$$

where $co_{c,t,s,r}$ is rate of crop-N offtake, $a_{ch,t,s,r}$ is the scalar, i.e. the area of crops destined for human consumption (*ch*), $livn_l$ is the livestock N-offtake per head of livestock $l_{l,t,s,r}$ by livestock type (l). The term $fx_{c,t,s,r}$ represents N fixation per hectare of crop $a_{c,t,s,r}$, $lu_{l,t,s,r}$ is total livestock units, $l_{l,t,s,r}$ livestock numbers and Nin_t is N contained within imported concentrate (i.e. cereals and beans). Imported compound feed (e.g. soy cake) is represented by $imp_{l,t,s,r}$ and $compn$, i.e. the total compound feed tonnage and the N content/tonne, based on feed values provided by Watson *et al.* (2010). The term $dp_{c,t,s,r}$ represents average atmospheric N deposition values, derived from national pollution data downloaded from the Centre for Ecology and Hydrology (CEH) website (http://www.pollutantdeposition.ceh.ac.uk/). N supply and offtake values for crops and livestock products were derived from Defra Fertiliser Recommendations (Defra, 2010) and the nutrient budgeting software PLANET (Dampney and Sagoo, 2008). To capture manure requirements for individual crops, a separate manure supply and demand constraint was applied within each region (see Appendix C).

6.3.12 Scenario testing

A base run of the model was produced, applying the data sources, assumptions and constraints described above, in order to generate a "best-guess" of what a wholly organic England and Wales would look like. The results of this base run were used as a comparator for additional scenarios in which parameters and constraints were adjusted to explore sensitivity of the scenario to changes to key assumptions. The scenarios and associated adjustments are summarised in Table 6.2 and explained below.

N fixation rate: As biological fixation by legumes is the main N input to organic systems, and reliable estimates of the amount of N fixed by different N-fixing crops under different conditions are difficult to obtain (Herridge *et al.*, 2008), two scenarios were run to explore the effect of higher and lower fixation rates, where the base run represents an 'average' fixation rate. The amounts of N fixed at these high and low rates were derived from Peoples *et al.* (2009), Schmidt *et al.* (1999) and Herridge *et al.* (2008). These altered N-fixation rates were also used to generate new crop yield estimates within NDICEA (see 'crop yields' section above).

Clover ley area: Organically managed arable land must be periodically diverted to fertility-building leys. This reduces the area that is cultivated over time compared with conventional systems. To explore the impacts of this, two scenarios were run with high and low clover-ley rates, with the average used for the base scenario.

Stocking rates: The effects of varying stocking rate constraints on livestock outputs were also assessed to capture intensive and non-intensive organic livestock production, using high/low stocking rate ranges based on data derived from AHDB Dairy (2012) and Moakes *et al.* (2012, 2014).

Fallow land: A significant area of fallow (non-productive) land was enforced in the base run reflecting average historic organic practice (up to 13% of the total area in the case of cropping farms was fallow). A separate scenario explored the impact of removing this constraint, i.e. allowing fallow land to be cultivated in order to reduce supply shortfalls.

Combined scenario: In a final scenario, two constraint settings were adjusted simultaneously. First, a new source of feed stocks (processing residue from imported cereals) was included in the livestock feed availability equation. Second, fallow areas were added to the cultivatable land area.

Results for each scenario were compared with three references: (i) the base run; (ii) the observed situation in 2010 under conventional agriculture as recorded in the June Survey of Agriculture (Defra, 2011; Welsh Government, 2011); and (iii) the projections of Jones and Crane (2009) for a wholly organic agriculture. The latter was undertaken for validation purposes.

We consider the 'combined' scenario to be the most likely outcome of 100% organic conversion as it does much to address the supply shortfalls seen in the base run. The results from this scenario were therefore used to assess the potential impacts of a 100% conversion on human nutrition. This was done by assessing the food outputs by each food group within a healthy eating framework developed in the UK, i.e. the Eatwell Plate (Macdiarmid *et al.*, 2011). This comparison addressed the question of whether the mix of products produced by a wholly organic agriculture is more closely aligned with the requirements of the Eatwell Plate than conventional agriculture, for example by supplying more fruits and vegetables than can currently be supplied by domestic sources.

Table 6.2. Scenarios assessed within the sensitivity analysis, defined in terms of their adjusted parameters

Scenario name	Parameters adjusted
Low N fix	Low crop yields and N fixation rates from NDICEA modelling
High N fix	High crop yields and N fixation rates from NDICEA modelling
Low Clover area	10% reduction in area of grass/clover leys as % of total utilisable area (UAA)
High Clover area	10% increase in area of grass/clover leys as % of total utilisable area (UAA)
High stocking rate	Upper/lower bounds on stocking rates per ha increased
Low Stocking rate	Upper/lower bounds on stocking rates per ha decreased
No fallow	Non-productive land (fallow) added to cultivatable area
Combined	Imported food residue added to livestock feed and fallow added to cultivatable area

6.4 Results

6.4.1 Cereals

Under all the organic scenarios, wheat and barley production was considerably reduced compared to the conventional non-organic baseline (Figure 6.4). Averaged across the scenarios, organic production was only 42% of the 2010 non-organic baseline production. For wheat, the greatest reductions were for the low stocking rate, high clover area and low-N fixation scenarios, due to combination of lower manure availability, cropland availability and crop yields. Reductions in output were less severe for barley, although the levels of production were more variable, ranging from 26% of the non-organic baseline under the low-N fixation scenario to 73% in the low clover scenario. Reductions in barley output were less severe for the low clover area, and high N fixation scenarios, as a result of both higher cropland areas and higher yields. The production of oats was relatively stable over the scenarios, reaching the upper limit of 150% of the baseline area in some scenarios. Production (and production areas) of oilseed rape (OSR) were also relatively consistent, in showing significant losses in all scenarios with an overall average production of 2.5% of the 2010 baseline (data not shown). Production estimates for wheat and barley are similar to those reported by Jones and Crane (2009), whereas the projections for beans and peas are higher, probably as a result of their increased representation in rotations. Oat production was much lower than reported by Jones and Crane (2009), probably as a result of the imposition of upper limits on production area in the OLUM.

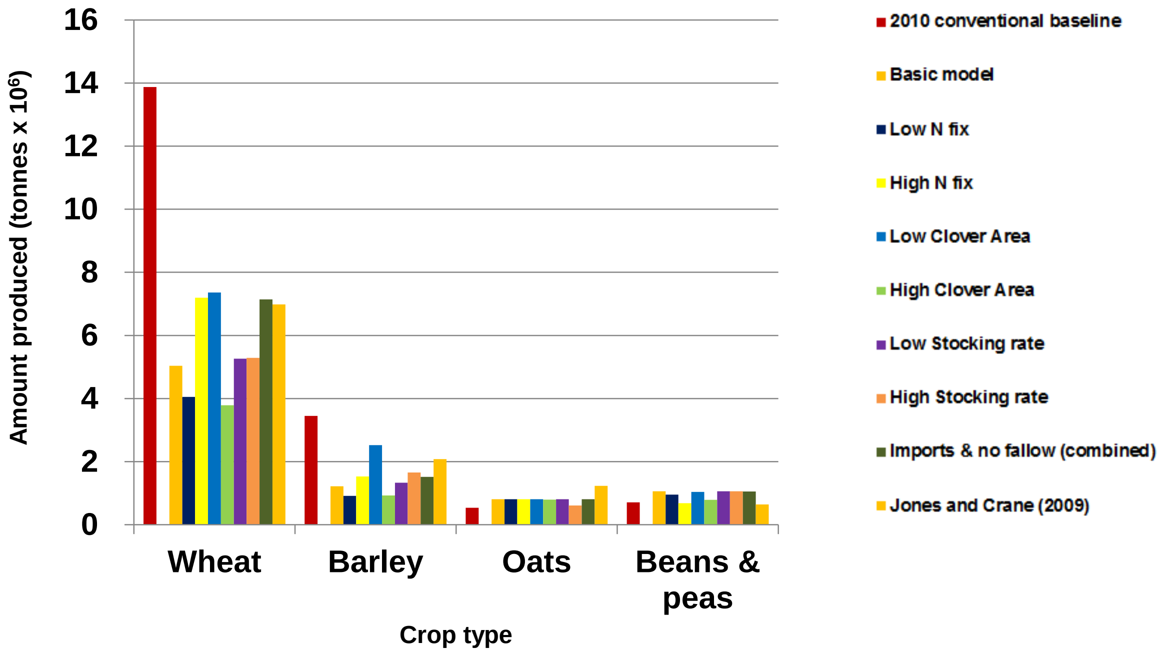

Figure 6.4. Production of arable crops in England and Wales under organic management scenarios compared to a 2010 conventional baseline and results from Jones and Crane (2009).

6.4.2 Other crops

Potato production was generally higher than in the non-organic baseline (Figure 6.5). This is reasonable because potatoes are common in organic rotations due to their beneficial effects on soil structure and for weed control. Potato production was less in the low-stocking rate scenario due to the lower livestock-manure-N availability. However, output volumes under all other scenarios exceeded the conventional baseline and the estimates of Jones and Crane (2009). Production of sugar beet was more variable than potatoes across the scenarios as the high-N offtake per hectare greatly affected the amount that could be produced under the low-N fixation and low stocking rate scenarios.

Brassica and protected vegetable crop production varied considerably across the scenarios with the highest production found under the high-N fixation scenario (Figure 6.6). With higher stocking rates production was reduced, as additional land and manure-N was required for feed crops. Production of root crops (onions, leeks, carrots) reached the upper constraints in most of the modelled scenarios, illustrating their relatively high energy values and resource-use efficiency (Carlsson-Kanyama *et al.*, 2003).

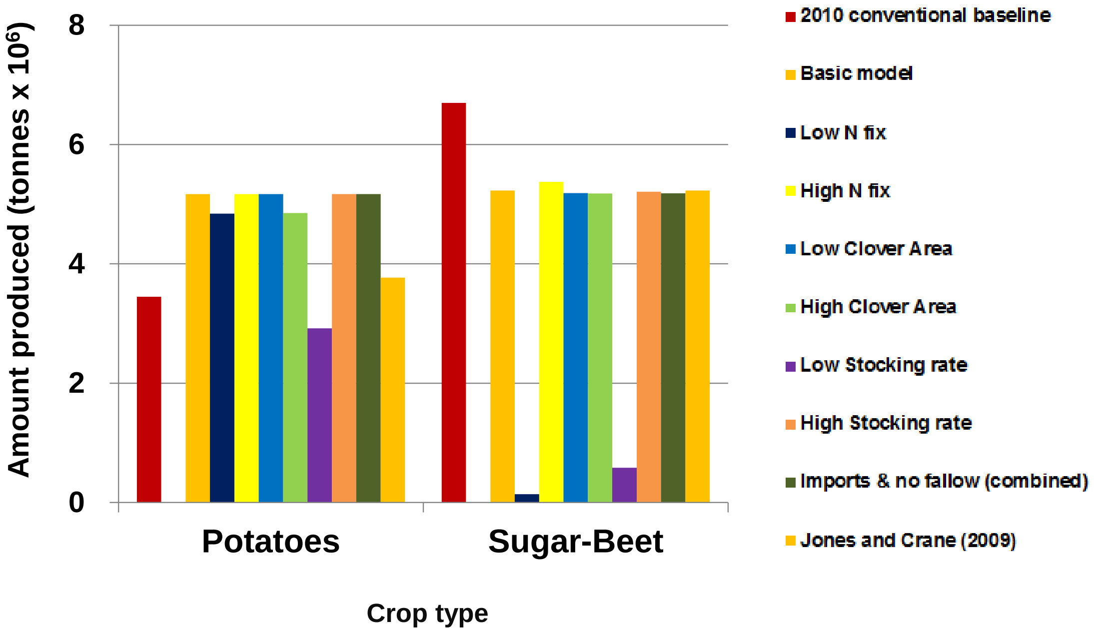

Figure 6.5. Production of potatoes and sugar beet in England and Wales under organic management scenarios compared to a 2010 conventional baseline and results from Jones and Crane (2009).

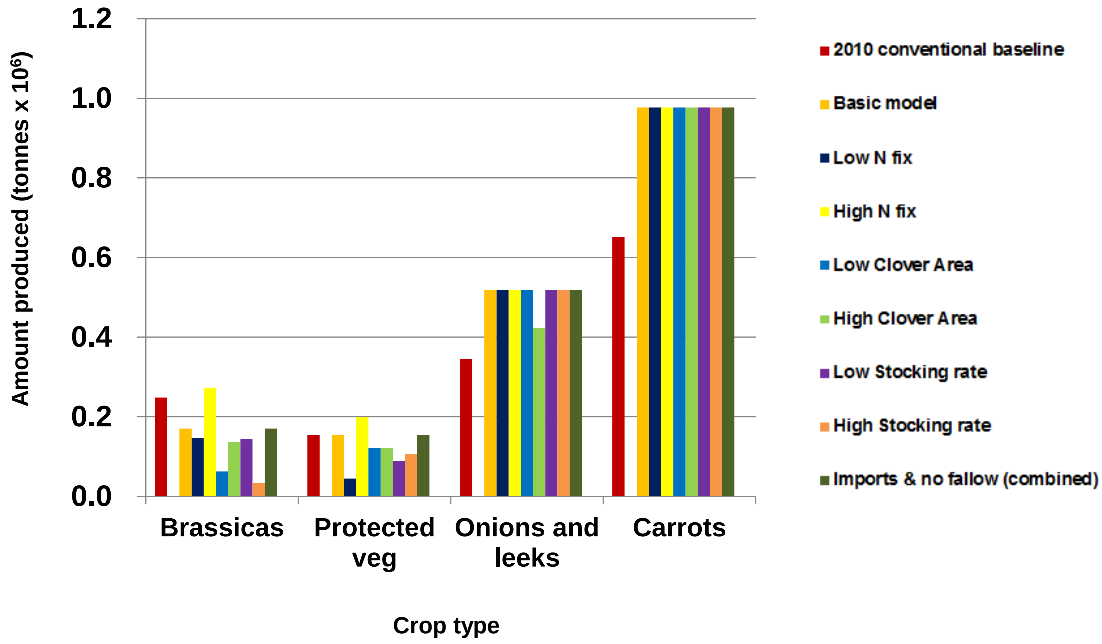

Figure 6.6. Production of vegetable crops in England and Wales under organic management scenarios compared to a 2010 conventional baseline.

6.4.3 Grazing livestock

Increases in beef cattle and sheep numbers, above the 2010 conventional baseline, were observed across all scenarios (Figure 6.7). This was particularly so for the "combined" scenario, due to increases in feed availability from recycled residues. The lowest rates of sheep meat production occurred under the low-stocking rate and low-N fixation scenarios, in the latter case due to lower cereal yields and the consequent reduced feed availability. Dairy cattle numbers and milk production were more sensitive to changes in N availability, cropping area, and cereal yield, due to a higher reliance on concentrate feeds compared to beef and sheep. Overall, milk production reached between 40% (low N fixation scenario) and 90% (high stocking rate scenario) of the 2010 conventional baseline. Despite the increase in beef and sheep livestock numbers, total carcass production for these livestock types was comparable to the conventional baseline as a result of longer finishing periods and lower carcass weights in organic systems (Figure 6.9).

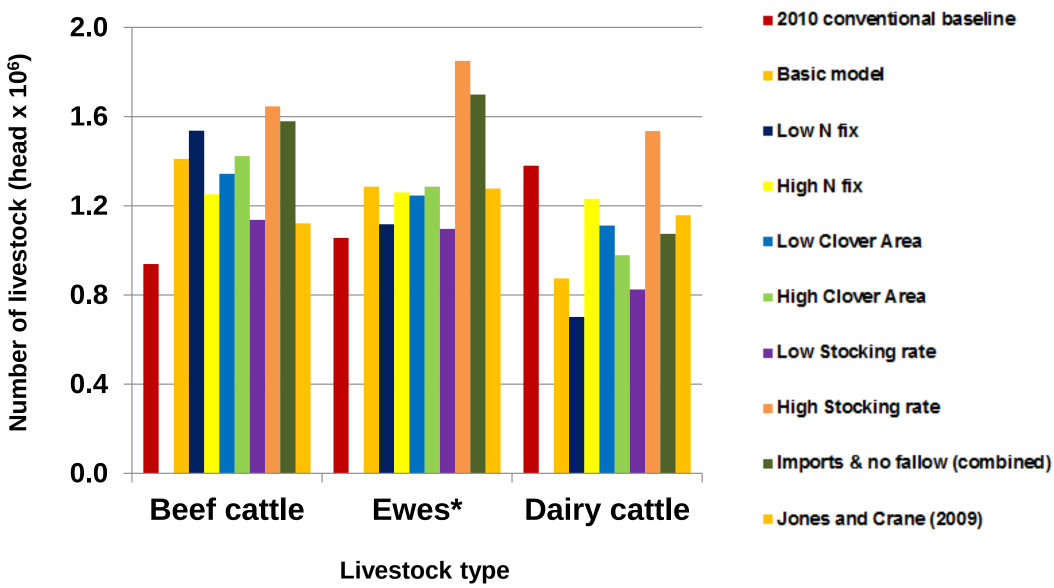

Figure 6.7. **Ruminant livestock numbers in England and Wales under organic management scenarios compared to 2010 conventional baseline and results from Jones and Crane (2009). * = head x 10.**

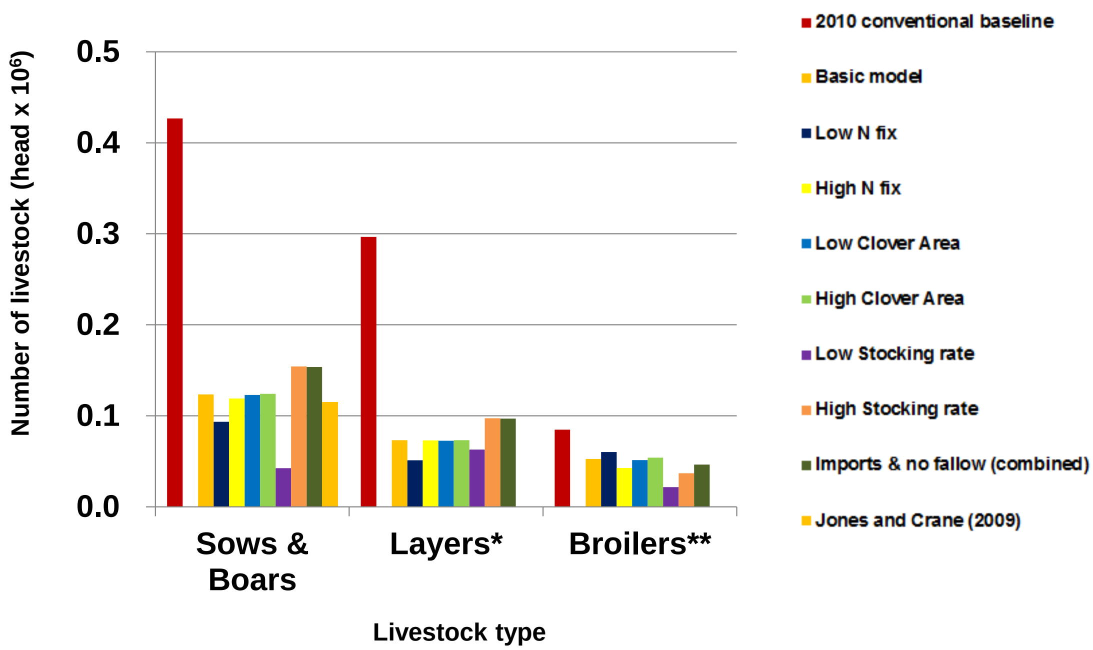

Figure 6.8. Monogastric livestock numbers and outputs in England and Wales under organic management scenarios compared to 2010 conventional baseline. * = head x 10^2 , ** = head x 10^3.

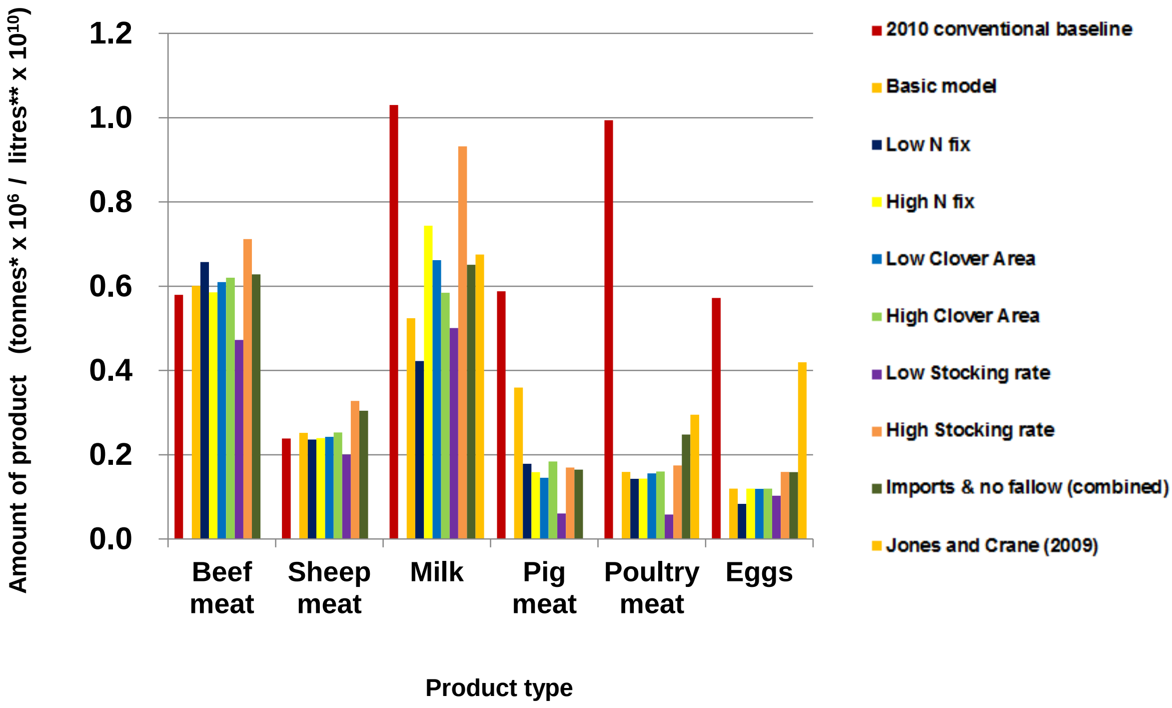

Figure 6.9. Output of livestock products in England and Wales under organic management scenarios compared to 2010 conventional baseline. * = meat and eggs, ** = milk.

6.4.4 Pigs and poultry

A major reduction in monogastric livestock production was predicted in all the organic scenarios compared to the conventional baseline (Figure 6.8 and Figure 6.9). Laying hen numbers were particularly affected, with total numbers reduced to 25–30% of the non-organic baseline, and considerably lower than the estimate by Jones and Crane (2009). The difference is due to limits on feed supply, as evidenced by the increase in monogastric numbers when dependence on home-grown feed supply was reduced under the Combined scenario. A similar effect is observed for pig production systems, although the values were closer to those projected by Jones and Crane (2009).

6.4.5 Metabolisable energy supply by food group

Conversion of the output volume results to output ME by Eatwell food group enables an assessment of the ability of a wholly organic agriculture in England and Wales to provide the food required by the populations of these countries (Figure 6.10 and Appendix D). The results show that fruit and vegetable production could almost match the 2010 conventional baseline levels under the Combined organic scenario (i.e. with increased feed availability and a reduced fallow area) with increases in outputs in eastern and south-west regions offsetting reductions in other regions. This illustrates the relatively small difference between organic and non-organic yields for field vegetables and the relatively small production areas required. However, the ME output of fruit (in particular apples and strawberries) was considerably less than the conventional baseline (data not shown), in part because of cosmetic standards within the retail sector (Smith *et al.*, 2015). The relatively high organic productivity seen in the fruit and vegetable food groups results from the high yields and outputs of ME per hectare for many of these crops (e.g. carrots and potatoes). The losses in output and food energy in starchy crops (e.g. wheat and barley) are a result of low yields, plus the requirement to divert land to clover/grass leys in arable rotations. Smaller reductions for this food group were found in western, livestock-dominated areas, which tend to have lower yields under conventional agriculture.

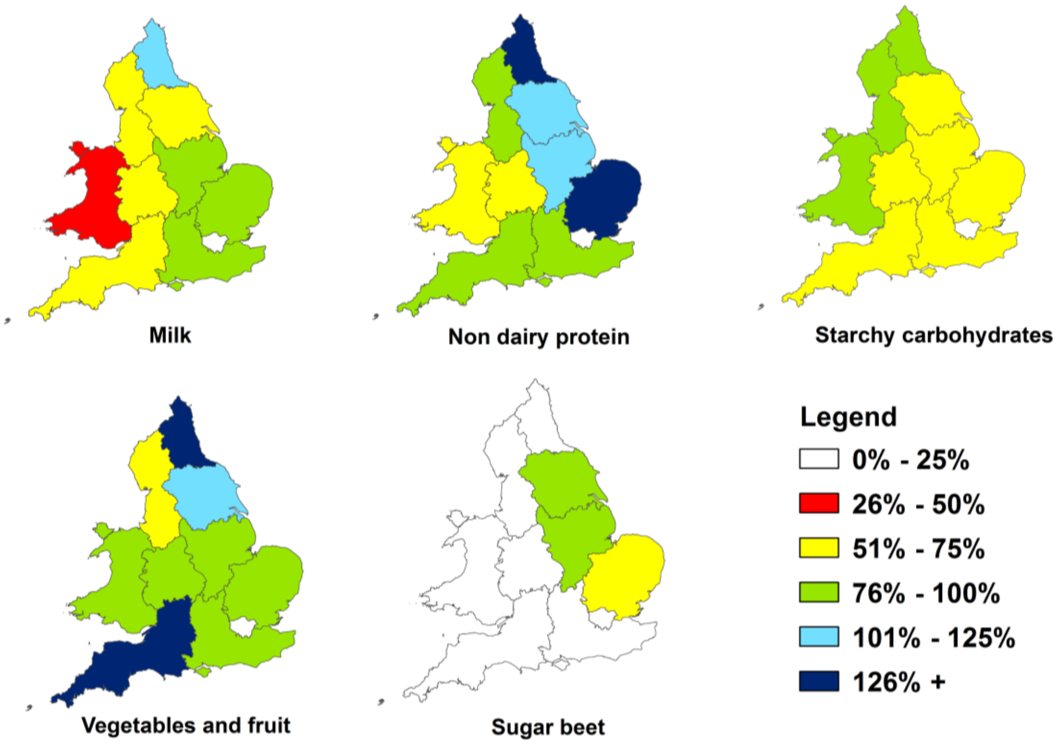

Figure 6.10. Food production (expressed by Eatwell group) in England and Wales under organic management as a percentage of a 2010 conventional baseline, expressed as total ME by NUTS 1 region (100% level = conventional production in 2010). The output by group refers to production only (e.g. wheat and potatoes in the case of starchy carbohydrates) as opposed to processed foods. Commodities allocated to each group within this study are shown in Appendix D.

Milk ME was substantially reduced under the Combined scenario, at just under two thirds of 2010 levels (see Appendix D) although introduction of dairy herds results in a small production increase in the eastern counties of England. In terms of total protein production, the reduction in meat and egg supply is somewhat offset by the increase in grazing livestock and peas and beans, although there is still an overall reduction in protein supply, in particular resulting from a decrease in poultry-meat and pork production under organic management.

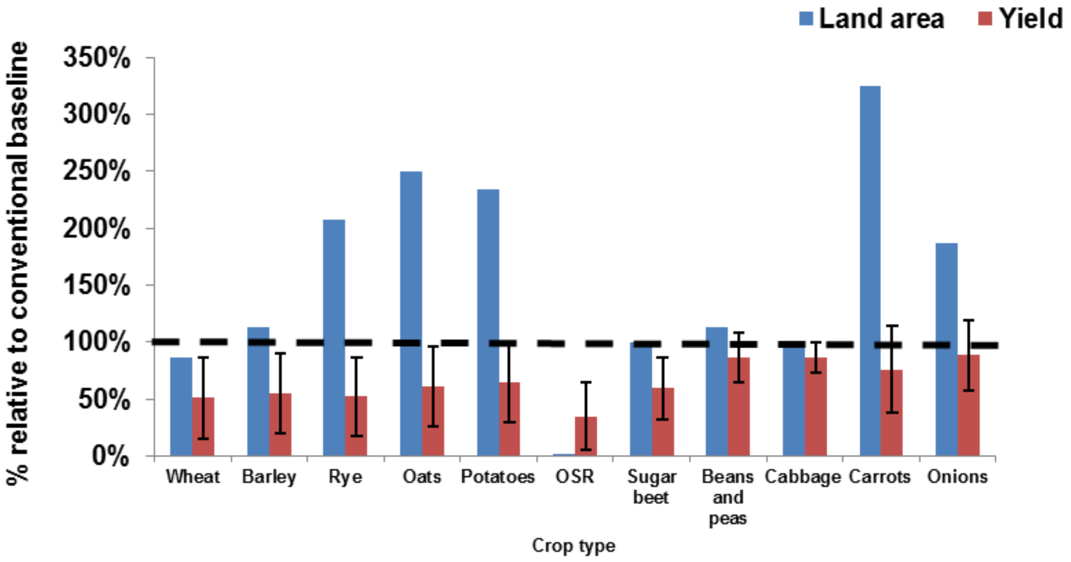

Figure 6.11. Crop area and yield under a 100% organic England and Wales agriculture, expressed as a percentage of the 2010 conventional baseline. Conventional yield data are from Nix (2009), the Farm Business Survey (2010) and Defra Horticultural Statistics (2010). Organic yield data are derived from the NDICEA-based adjustments and published sources described above. OSR = oilseed rape. Error bars indicate standard deviation.

Figure 6.11 shows that the decrease in wheat production projected by the Combined scenario is a result of reductions in both area cultivated and crop yield (organic yields are only 51% of non-organic production). Oats and rye have smaller yield losses under organic production, but the projected increases in production under organic scenarios are largely due to increases in the area cultivated. The low productivity of organic oilseed rape is compounded by a much smaller cultivated area (which itself probably results from the low yield). The increase in bean production under the organic scenario is a result of the increase in production area, this being driven by the need to maintain fertility (as reflected in the rotation constraints). It is also the case that such N-fixing crops produce yields under organic management very close to conventional. The production area of potatoes is also substantially increased under the organic scenario, whilst sugar beet areas hit the upper constraint on production area.

6.5 Discussion

6.5.1 Reprise of modelling outcomes and comparison to outputs from previous work

The results showed that converting agriculture in England and Wales to organic management would result in a major drop in food production, with total food output (as metabolisable energy, ME) falling to 64% of non-organic baseline levels (Appendix D). The reductions in crop output would be most severe for major cereal crops, sugar beet and oilseed rape, as a result of reduced yields, the need to divert land to fertility building leys, pest, disease and weed susceptibility and high N demands in the spring (Schneeberger *et al.*, 2002; Tzilivakis *et al.*, 2005; Valantin-Morison and Meynard, 2008).

Carrot yields are also lower under organic management, due to susceptibility to weeds and carrot fly attack. Despite this, the relative efficiency of this crop, in terms of energy output relative to N requirements, resulted in a potential overproduction compared to 2010 levels (see Figure 6.6) and a substantial increase in the production area. Vegetable, potato and leguminous crop production could meet or exceed the non-organic baseline due to smaller yield losses for these crops within organic systems and over-representation of these crops in organic rotations. Beef and sheep numbers could also increase overall as a result of increased stocking levels in arable dominated areas, whereas poultry and pig production and output would decrease, due to the requirement for more extensive production practices under organic certification. Dairy cattle numbers and total milk production would also decrease as a result of lower stocking rates and lower milk yields under organic management.

The similarity of the results reported here to outcomes from the study by Jones and Crane (2009), which used a different approach, suggest that the estimates are robust and therefore realistic as far as such an extreme scenario can be predicted. There are some interesting areas of divergence however, in particular the projection, in this study, of much lower volumes of egg production. This divergence has been caused by limits on domestic feed availability present in the OLUM model, reflecting constraints on the amount of imported feed and upper constraints on the stocking rate per hectare. Estimates of total wheat and barley production were also considerably lower than Jones and Crane (2009), and it is possible that Jones and Crane (2009) overestimate organic yields of major cereals due to sampling error, i.e. the Farm Business Survey, which was the sole-source of the organic yield data used in the 2009 study, is known to over-represent larger, more commercial farms (Jones and Crane, 2009). Conversely, pea and bean production estimates are higher in this study, driven by rotational requirements (nearly all of the rotations applied included a legume crop). Production of sugar beet was comparable with Jones and Crane (2009), except in cases of reduced N supply (i.e. the “low stocking rate” and “low N-fixation” scenarios). Milk production levels were similar to the 2009 study, although production volumes exceeded those projected in Jones and Crane (2009) under the higher stocking rate scenario.

6.5.2 Implications for national diets

The results suggest that a widespread conversion to organic farming would have major implications for domestic food supply. Without a fundamental change in consumption habits, these losses would require, in compensation, considerable increases in imports, implying a corresponding expansion of cultivated agricultural land overseas.

An analysis of the extent to which diets would need to change in order to accommodate reduced supply of important commodities from organic production is beyond the scope of this study. However, some qualitative conclusions can be drawn. The results suggest dietary changes would need to include a reduction in the consumption of poultry meat and eggs, increased consumption of beef, lamb and non-meat protein (in particular beans and peas) and increased vegetable consumption in some regions. Although an increase in bean, pea and vegetable consumption would be relatively consistent with the changes to western diets currently being recommended by health professionals, an increase in red meat and a drop in poultry meat consumption could represent a conflict. A requirement for a reduction in wheat consumption could also present major difficulties, and this would need to be compensated for by an increased consumption of other forms of domestically-produced starchy carbohydrate (e.g. potatoes and oats), or increased imports of crops like maize and rice.

It should also be noted that fish supply by catching or farming was not addressed, but increasing fish consumption in the UK, especially oily fish, is recommended (Macdiarmid *et al.*, 2011).

6.5.3 How feasible are dietary changes on this scale?

Dietary changes on this scale would be difficult to achieve in a free market, particularly in view of the UK's dependence on wheat as a staple, the lack of policy mechanisms to drive this (especially mechanisms that do not add to total food costs) and the lack of political will to invest the time and resources needed to transition to healthier and more sustainable diets (Wellesley *et al.*, 2015). Encouraging greater consumption of vegetable crops is likely to require a significant overhaul of policy support measures in the UK and Europe, which has in the past tended to promote the (over) production of meat, sugar and dairy products, thereby driving down market prices, leading to over consumption, particularly in low income households (Bailey *et al.*, 2016; Birt, 2007). With the recent decision to leave the European Union, the UK has an opportunity to change the balance of support for agriculture and reduce the environmental and health impacts of the food system, whilst providing additional jobs in a labour-intensive sector (Schoen and Lang, 2016).

There is also some evidence to suggest that typical 'organic consumers' are directed towards more sustainable food choices, exhibiting preferences for fresh vegetable consumption and vegetarian food (Pelletier *et al.*, 2013). A shift to organic consumption habits would therefore be likely to lessen the environmental impacts of widespread organic conversion (Baroni *et al.*, 2006). Moves in this direction could also help to encourage nutritionally balanced diets and the associated health benefits (Macdiarmid *et al.*, 2011). The increased costs to consumers associated with organic production and consumption could present a major challenge, however, particularly in view of the current lack of willingness to pay more for sustainable diets (von Koerber *et al.*, 2017).

The results of this study also highlight the importance of food waste reduction in achieving a more sustainable supply of food. With over 27% of the food purchased in the UK wasted in 2015 there are still considerable opportunities for offsetting supply losses in this area through improving: management practices in the retail sector (e.g. avoiding overstocking); technological innovations (e.g. smart-fridges); and educating consumers (e.g. the Love Food Hate Waste Campaign introduced by the Waste and Resources Action Programme (WRAP) in 2007 (Priefer *et al.*, 2013). If such measures could be implemented on a wider scale, they would help to reduce overall food demand, thereby reducing the significance of the supply shortfalls projected under extensive organic conversion and in so doing allow for the wider adoption of this agricultural system, accruing the lower resource-use benefits that would accompany it.

6.5.4 Other ways to counter production volume shortfalls

Another possible solution to under-supply of major commodities might be to bring more land into arable cultivation, as was done on a large scale in the UK during the Second World War. However, converting non-productive land (e.g. parklands) or non-arable land, such as woodland or low input permanent grassland) to arable production would result changes to landscape character, loss of amenity and potentially severe environmental impacts. Such a move would also run counter to multiple environmental protection policy objectives as set-out in the UK Climate Change Act, the UNFCCC Paris Climate Change agreement, the UN Convention on Biodiversity and the EU Biodiversity Strategy. Some increased production from urban farms and gardens could be envisaged.

A relaxation of organic standards might also be envisioned (such as an increase in permissible stocking rates and maximum flock sizes) to increase commodity supply under a 100% organic scenario. Reducing the area of grass/clover ley within cropping systems could also allow for greater crop production, as illustrated within the “low clover area” scenario. Although such a shift may have positive impacts in terms of land availability for the productive phase of rotations in the short term, it is likely to ultimately result in decreased N availability and increased occurrence of pests, diseases and weeds, as use of grass/clover-leys in organic systems is the primary method of controlling these factors (Lampkin, 2002). A balance would therefore be required in terms of the optimum amount of ley relative to the cropping phase, although this is likely to vary with climate, soil and other conditions, such as labour availability post-harvest for ley establishment. Difficulties associated with the prediction of N supply from grass/clover leys can also present major challenges, in particular for stockless systems, which rely on biological fixation for the supply of N and can struggle to maintain a positive N balance over the course of a rotation, particularly in wetter areas and on lighter soils (Smith *et al.*, 2016). In addition, reducing the grass/clover ley percentage in organic rotations may offset some of the purported benefits of organic approaches in terms of enhanced C sequestration and biodiversity (Lampkin *et al.*, 2015). P supply could also become critical in due course and a change in organic standards to permit the use of sewage sludge would promote the circular economy and produce a valuable supply of P and N, along with organic matter.

6.5.5 Future research and methodological reflection

Although the modelling approach used in this study extends the approaches deployed by Jones and Crane (2009), i.e. by increasing the range of factors taken into consideration in estimating production volumes (e.g. concerning yield variation by land class and constraints on livestock feed demand), the approach is still somewhat restricted. The primary limitation is that the objective function of the OLUM is maximisation of food production, as this does not fully reflect the diverse and multifaceted business goals of farmers. An economic approach to the upscaling of organic agriculture, i.e. the use of a profit maximising model, may yield considerably different results, although the input costs and price differentials under a 100% organic scenario are likely to be highly spurious given uncertainty over product prices and changes to the costs of inputs in such an extreme situation. It should also be considered that in this modelling exercise organic systems in their *current form* were scaled up to the national level, but constrained by biophysical factors. Under a 100% organic scenario it may be reasonable to expect a significant restructuring of the industry to avoid some of the supply shortfalls observed here. Although it is likely that the broad structure of agriculture in England and Wales will remain the same, due to immutable agronomic constraints (e.g. cropping dominating in the eastern areas and ruminant livestock dominating in the west) there will be some loss of specialisation, i.e. a shift towards greater arable production in livestock dominated areas (e.g. Wales and the south west of England) and expanded ruminant livestock production in the Eastern Counties, where specialist arable farms currently predominate

It is also quite possible that a widespread switch to organic methods would have a much greater impact on food production than estimated within this study. For example the approach employed here assumed that the organic industry would maintain the current (conventional) mix of farm-types by region, however the current trend within organic agriculture in the UK is for a high proportion of farms, including arable farms, to host ruminant livestock, i.e. producing beef, lamb and dairy products (Defra, 2015). If this arrangement were scaled up to the national level the impacts on food security in the UK could be even more severe, at least for arable production, although the output of beef and sheep would be likely to increase dramatically. In addition, in the OLUM model, stockless production was permitted *ad libitum* on farm types dominated by cropping (i.e. specialist cereal and general cropping farms), whereas stockless systems are currently rare in UK organic production (economically and agronomically it can be difficult to justify the maintenance of ley/arable rotation without livestock). One way in which stockless arable farms can make economically rational use of grass and clover leys is to use the forage produced for other purposes. For example grass and clover can be an efficient feedstock for anaerobic digestion (AD) plants, if these can be situated on the farms where the feedstock is produced, or at a reasonable distance to them (Halberg *et al.*, 2008). The economic impact of the modelled scenarios projected here is beyond the scope of this study to assess. However, such approaches could contribute to making the application of stockless ley/arable systems more viable on a wider scale. The digestate fertiliser provided by the wider application of AD on arable farms could also help to enhance organic crop yields by providing a source of readily-available N to meet crop requirements at times of peak demand (Stinner *et al.*, 2008).

In future developments of the OLUM model it would be possible to construct a scenario where forage can be used for heat and/or electricity generation. Careful consideration would of course have to be given to constraints limiting the extent to which the model can deploy this option, reflecting the fact that the development of farm-scale AD has, and continues to be, slow in the UK as a result of perceived risks, and relatively poor economic returns for smaller scale plants (Jones and Salter, 2013). It is therefore likely that the use of forage for feedstock, on a large scale, would depend on the application of centralised AD plants with a number of organic farms providing feedstocks from a distance.

The results from this study also illustrate the dependence of organic systems on N supplied within the farming system, in particular on the supply of manure. In this study the assumption was made that manure would not be transported outside a given region. However, it would be reasonable to expect that transfer might occur over larger distances (e.g. from livestock-dominated areas in the south west of England to arable areas in the east) although transport costs, increased disease risks and odour may make long-distance transport infeasible (Sims *et al.*, 2005). Some successes have been achieved in installing central manure processing plants in the Netherlands, to help deal with N surpluses at a local level, although the financial viability of such systems has been difficult to maintain, even in cases where the final product is 'dewatered' to facilitate transport (Zwart, 2015).

The approach taken to reflecting N availability and its influence on yields in the OLUM model is also fairly rudimentary, i.e. through focussing on N availability under a limited range of environmental conditions (i.e. soil type and rainfall). A more complex model of organic crop yields could take account of factors such as pests and diseases, water stress and annual variations in areas of grass/clover ley, whether caused by environmental (e.g. average temperatures within each region) or economic (e.g. availability of labour) factors. The effect of climate change on the production scenarios could also be considered, for example, allowing for the predicted northward expansion of sunflower production and possibly soybean into the UK (Olesen and Bindi, 2002). Expanding the model to consider a broader range of nutritional requirements and supply values for crop and livestock products (e.g. iron, calcium) and/or environmental criteria (e.g. greenhouse gas emissions per tonne) could also allow for an increased range of scenarios to be modelled, for example to estimate the optimum balance between healthy food choices and environmental sustainability, as reported in Macdiarmid *et al.* (2011). The OLUM provides an invaluable framework for the assessment of such scenarios by providing a model that emulates the current national structure of the agricultural industry and current practices on typical farms.

6.6 Conclusion

In summary, the results from our study suggest the impact of full conversion to organic farming on food production in England and Wales would be severe. The losses would be greater for some commodities (e.g. cereals, oilseeds, monogastric livestock) than others (e.g. vegetables and milk). The relative similarity of organic vegetable yields to conventional make this the most likely cropping sector to be able to sustain widespread adoption of organic practices. The results also suggest that certain organic practices could be expanded within some non-organic systems to improve resource use efficiency, without jeopardising production. This could include greater use of clover in grassland and/or introducing livestock to field vegetable cropping systems. To lessen the need for large-scale compensating increases in imports, significant changes in diet would be required (e.g. reducing pig and poultry meat consumption). Efforts to reduce production losses and losses in the food chain would also be required.

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CHAPTER 7. THE ENVIRONMENTAL IMPACTS OF A 100% CONVERSION TO ORGANIC FARMING IN ENGLAND AND WALES

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Research methods applied	Life Cycle Assessment Land occupation assessments Estimation of soil carbon sequestration in arable soils Sensitivity analysis

7.1 Abstract

The consequences for net greenhouse gas (GHG) emissions of a widespread conversion to organic farming practices are largely unknown. We assessed the impact on GHG emissions of a 100% shift to organic food production in England and Wales using Life Cycle Assessment methods and a linear programming model. We included both the GHG emissions associated with the diversion of land overseas to supplying shortfalls in UK supply under organic production, and the offset that could be achieved through soil carbon sequestration benefits attributed to organic systems.

Major reductions in food production were found under fully organic production, in particular for wheat, barley, oilseed rape, pork, eggs and poultry meat, though production of potatoes, oats, minor cereals and ruminant meat increased. Direct GHG emissions were less under organic management, despite increased emissions per tonne for some products. However, when the land use change impacts associated with increased imports were included, the GHG savings that could be achieved through widespread organic conversion were offset, leading to higher GHG emissions overall. The offsetting effect of inclusion of the C sequestration benefits of organic management varied greatly depending on the assumptions made.

We found that organic farming practices can reduce the GHG emissions and fossil energy use associated with the production of many agricultural commodities. However, a widespread conversion would necessitate a considerable change in the national diet to avoid a major compensatory expansion in food production overseas to redress supply shortfalls in the UK market, and the GHG emissions associated with land use change.

7.2 Introduction

Organic farming systems aim to have a lower greenhouse gas impact than conventional through a lower production intensity, less use of manufactured inputs, and greater soil organic carbon (SOC) contents (Lampkin *et al.*, 2015). It is suggested that organic practices could contribute significantly to achieving national GHG reduction targets (e.g. Scotland's Organic Action Plan). However, as far as we are aware there have been no rigorous assessments of the validity of this suggestion, covering the main agricultural GHGs: methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) from both primary and secondary sources.

Government recommendations and support policies promoting organic practices have to some extent been informed by studies using Life Cycle Assessment (LCA) approaches (e.g. Williams *et al.*, 2006). LCA offers a robust, comprehensive and easily understood method for the assessment of the environmental impact of different production systems and individual management practices (Notarnicola *et al.*, 2017). Through the adoption of a whole supply chain approach, LCA methods avoid the omission of key environmental externalities in sustainability assessments. LCA methods are underpinned by international standards (ISO, 2006), which helps ensure comparability between studies, and they can be used to assess a wide range of scenarios, for example, changing cropping patterns, adjusting diets for livestock, or bringing more land into agricultural production, across a range of scales (Lehuger *et al.*, 2009; Vázquez-Rowe *et al.*, 2014).

LCA studies exploring the environmental impacts of organic cropping have yielded mixed outcomes, arising from differences in assessment approaches (in particular the system boundaries) and data sources, and variation in environmental factors and/or typical practices, between countries or regions (Meier *et al.*, 2015). For example, Williams *et al.* (2006) found that most organic field cropping systems in England generate similar or greater GHG emissions per tonne compared to conventional systems, with lower yields and increased rates of nitrate leaching offsetting lower use of inputs. Conversely, a Swiss study reported significantly lower GHG emissions per tonne of organic product (Nemecek *et al.*, 2011). These lower GHG estimates resulted from a focus on outputs from the system as a whole (i.e. the entire crop rotation) and less-intensive modes of production compared to the Williams *et al.* (2006) study.

Only a few studies have applied LCA methods to the aggregate impacts of a widespread conversion to organic farm management. The most recent attempt was made by Audsley *et al.* (2009) who combined national data on commodity production, processing, distribution, retail and trade with the results from the Cranfield University LCA model (Williams *et al.*, 2006). The Audsley *et al.* (2009)

study compared a ‘baseline’ LCA assessment, reflecting observed, real-world consumption patterns, with a range of scenarios, one of which was the assumption of a transition to 100% organic diets in the UK. This organic scenario built on a study by Jones and Crane (2009) in which the production impacts of a 100% conversion to organic agriculture in England and Wales were estimated, through an application of data on organic yields, crop areas and livestock numbers from the Farm Business Survey (FBS). The results of Audsley *et al.* (2009) indicated that a switch to organic consumption in the UK could result in a GHG emission reduction of about 8% in terms of food production, but the GHGs associated with the additional overseas Land Use Change (LUC) required to meet UK supply shortfalls were not considered.

The study reported here builds on the assessments by Jones and Crane (2009) and Audsley *et al.* (2009) by accounting for (a) limits to organic production imposed by the supply of livestock feed and available N; (b) the GHG impact of overseas land use changes associated with increased food-imports under organic production, and (c) the GHG offset potential of soil carbon sequestration under organic production. This study therefore provides an updated and more-comprehensive assessment of the potential land use, food production and GHG impacts of an up-scaling of organic agriculture to achieve 100% coverage.

7.3 Method

A combination of land-use modelling and Life Cycle Assessment were used to assess how total GHG emissions would change under conversion to a 100% organic agriculture in England and Wales, as described in the following sections.

7.3.1 The OLUM farm type model

The OLUM (Optimal Land Use Model) model is described in detail by Smith et al. (2017). It includes a suite of activities that represent current organic practices within a range of farm types[12] across the entire agricultural land-base in England and

[12] The farm types contained within the OLUM model were based on the Defra Robust Farm Types i.e. specialist cropping, mixed arable and livestock, specialist dairy, lowland grazing livestock, Less Favoured Area (LFA) grazing livestock, pigs and poultry, other

Wales. It was constructed using the programming platform GAMS[13] and took a linear programming (LP) approach, where an objective function is maximised, subject to certain constraints (i.e. resource availabilities, defined as equalities or inequalities). The basic structure of the model is given in Equation 1:

$$\mathbf{Z} = \sum_{ij=0}^{n} \boldsymbol{C_{ij}} \cdot \boldsymbol{x_{ij}} \tag{1}$$

$$\text{subject to}$$

$$\boldsymbol{R}\mathbf{x}(\mathbf{ij}) \leq \mathbf{b};\ \mathbf{x}(\mathbf{ij}) \geq \mathbf{0}$$

where:

> $\boldsymbol{Z}$ is the objective variable (i.e. the value to be maximised, expressed as the total metabolisable energy (ME) of all food crops and livestock products;
>
> $\boldsymbol{C_{ij}}$ is the ME output of individual organic agricultural products summed over i product types on j soil/rain classes;
>
> $\boldsymbol{x_{ij}}$ is a scalar for the agricultural activities (i.e. crop areas/livestock numbers);
>
> $\boldsymbol{Rx_{ij}}$ represents the input and resource requirements (R) associated with the agricultural activities (x_{ij}); and
>
> ***b*** is the resource endowment and input availability vector (e.g. land by soil and rainfall class).

Within each farm type, the set of crop and livestock production activities available were fixed, as evidence suggests that the dominant agricultural activity (e.g. dairy farming) will usually stay in place post conversion to organic management, due to existing farm infrastructure and local conditions (Howlett *et al.*, 2002). However, these activities could be individually expanded and contracted endogenously. The land areas under each of these farm types was fixed, reflecting the areal coverage of their conventional equivalents recorded in the June Survey of Agriculture in 2010

[13] General Algebraic Modelling System (GAMS). GAMS Development Corporation. http://www.gams.com/http://www.gams.com

(Defra, 2011). A number of logical constraints were applied within the model to reflect: the availability of land within the various soil/rainfall classes; maximum permissible area of crop groups (e.g. cereals, root crops) reflecting rotational constraints and upper limits on the total output of each crop, set at 150% of the current supply, following an assumption that further increases could not be absorbed by the market.

Rotational N availability limits were also imposed, as determined by crop and livestock-product offtake (from the land), N supply from various sources, such as biological fixation, imported feed and atmospheric deposition, as well as manure-N availability within each region. Livestock numbers and associated product output volumes were constrained by feed availability, maximum and minimum stocking densities.

The OLUM was run to produce a 'best estimate' of what a fully organic agriculture sector would look like. To ensure that the results from the base run were reasonable, outputs were compared to the real-world distribution of (conventional) production in 2010, derived from a range of industry sources specified in Appendix E, and to results from a previous study on the production impacts of a switch to organic farming in England and Wales (Jones and Crane, 2009). The environmental impacts of the organic production scenario were then assessed through an application of the Cranfield Agri-LCA models (Williams *et al.*, 2006).

7.3.2 The Agri-LCA models

The Agri-LCA models are stand-alone models that were designed to estimate GHG emissions from different agricultural systems in England and Wales, under various assumptions (Williams *et al.*, 2006). Results from earlier emissions analyses using these models, i.e. based on the current combination of agricultural systems in England and Wales, were used to create a comparator to assess the relative performance of the organic conversion scenario projected in the current study. To achieve this, estimates of the GHG emissions and fossil energy use associated with each tonne of conventional product reported in Williams *et al.* (2006) were combined with national data on levels of production to provide an estimate of the impact of conventional production in 2010. To assess the GHG implications of the organic scenario, the following components of the Agri-LCA models were adjusted to better reflect organic agriculture:

1. Crop and grassland yields
2. Crop cultivation practices and manure/compost application rates
3. Crop and grassland areas by soil and rainfall type
4. Livestock productivity and mortality rates
5. Livestock diet compositions

Crop yield, cultivation and manure application data were required for twelve main crops: wheat, barley, rye, oats, potatoes, oilseed rape, sugar beet, beans and peas, cabbage, carrots, onions and forage maize (i.e. covering 98% of the cultivated land in England and Wales, Defra, 2011b). Data sources are given in Appendix E. Crop and grassland areas under each of sixteen soil and rainfall classes were derived from the OLUM results. The adjusted crop areas, by each soil and rainfall class, were used to adjust the N_2O and CO_2 impacts of organic management. The functional units used in the LCA analysis were tonnes of marketed crop-product.

Organic animal production data were drawn from a range of industry sources to define by livestock type: daily liveweight gain, annual fat-corrected milk yield, and feed conversion ratios (Appendix E). Data were also obtained on the composition of livestock diets, stocking rates per hectare and the proportion of livestock on upland and lowland land (Appendix E). These values were applied within the Agri-LCA, ensuring that feed intake met the metabolisable energy demand of livestock. Nitrogen excretion from livestock was derived from mass balances. Compound feed composition data were also applied to determine embedded impacts of feed production overseas. Direct CH_4 emissions were calculated as a function of dry matter intake (scaled in proportion to the forage dry matter intake) liveweight and milk yields. The livestock assessments within the Agri-LCA focussed on six commodities: eggs, milk, sheep, beef, pig and poultry meat. Meat outputs were defined in terms of total dressed carcass weight (tonnes), eggs by weight (tonnes) and milk output as fat-corrected litres (Williams *et al.*, 2006).

7.3.3 System boundaries and allocation of environmental burdens

The downstream system boundary applied in the LCA was the farm gate, i.e. only inputs consumed during farm-based processes were considered. The GHG emissions associated with downstream activities – such as distribution, consumption and disposal of products produced on the farm – were not included. Some on-farm processing, such as grain drying, milk cooling and potato storage, were included in the total impact assessment, as these operations were considered to be part of the on-farm production process (Williams *et al.*, 2006). Allocation of the environmental burdens associated with fossil energy use, including GHG emissions, was achieved by economic value with respect to disparate outputs such as grain and straw and by system expansion with regard to manure (i.e. the manufactured N fertiliser avoided was discounted from the environmental burdens associated with non-organic crops). Emission factors were derived from IPCC 2006 estimates and total emissions converted to CO_2 equivalents using 100-year global warming potentials.

7.3.4 Carbon sequestration estimates

Annual topsoil carbon sequestration estimates under organically-managed land were made following Gattinger *et al.* (2012), who completed a pairwise meta-analysis of 74 studies and found significantly higher SOC concentrations and sequestration rates in organic systems, compared to conventional. Gattinger *et al.* (2012) noted that greater use of compost and manures/slurries in organic farming systems, together with longer and more diverse crop rotations, were important reasons for the difference, although not all the studies identified these drivers and some found lower rates of carbon sequestration in soils for zero-net-input systems. Given the uncertainties associated with the prediction of soil carbon sequestration rates (Smith, 2004), no single figure is appropriate and so we compared the three annual sequestration rates reported in Gattinger *et al.* (2012):

- Low – the rate reported for those zero-net-input systems where bulk densities and inputs were recorded: 0.07 Mg C ha^{-1} yr^{-1}
- Medium – the rate over all zero-net-input systems: 0.27 and 0.45 Mg C ha^{-1} yr^{-1}

- High – the rate for all studies that included sufficient data (i.e. peer-reviewed pair-wise farming system comparisons reporting data on SOC concentrations, where organic farming practices were applied for at-least 3 consecutive years): 0.45 Mg C ha^{-1} yr^{-1}

We highlight that the above gains will be time-limited, as any given soil will have a finite capacity to accumulate C and a new steady state will be reached over time (Powlson *et al.*, 2011). The sequestration rates therefore provide an estimate of the GHG offset potential that could be achieved in the early-years (i.e. < 20) following a conversion to organic methods (more in the discussion section). Sequestration rates in established swards of permanent pasture or rough grazing were also assumed to be zero under the assumption that these sites will already have reached a steady state (Smith, 2004).

7.3.5 Imports and exports

Quantities of food imported into England and Wales were included in the LCA to assess the total impacts of full organic conversion. We considered that any shortfall in supply from organic agriculture compared with historic conventional agriculture would need to be compensated by increased imports of organically produced commodities from overseas. Using data from a range of industry sources (Appendix E) the required amounts of imported product were allocated to particular global regions based on the historic regions of origin of conventional imports (Hess *et al.*, 2015). The GHG and energy-use associated with the transport of imports to England and Wales was determined by multiplying the total volume of imports by GHG and fossil energy use coefficients derived from Hess *et al.* (2015). Transport burdens for imported sugar and sheep meat were derived from Plassman *et al.* (2010) and Webb *et al.* (2013) respectively.

Estimates of the overseas land-use required to produce these imports were derived through average regional yield data from Eurostat, plus a recent meta-analysis on organic crop yields (de Ponti *et al.*, 2012) and from results of an LCA for milling wheat grown in Canada (Pelletier *et al.*, 2008). Land requirements to generate each tonne of imported livestock-product were derived from the Agri-LCA (Williams *et al.*,

2006) and recent studies on the environmental burdens of imported lamb from New Zealand (Barber and Lucock, 2006; Webb *et al.*, 2013). The additional land required for the supply of organic crops and livestock products was calculated as the difference between the amount of land required for imports in the non-organic baseline (based on the non-organic land-use values in Table 7.1) and the total amount required under each organic scenario.

Table 7.1: Land use per tonne values applied in this study to calculate overseas land requirements for imports. Note that the land use requirements refer only to the *non-forage* component of the diet

	Overseas land use requirement (ha per tonne $^{-1}$) - non-organic (Williams *et al.* 2006)	**Overseas land use requirement (ha per tonne^{-1}) - organic (Williams *et al.* 2006)**	**Overseas land use requirement (ha per tonne^{-1}) – organic- other literature sources**
Pork (dressed carcass)	0.7	1.3	1.0
Poultry (dressed carcass)	0.6	1.4	2.5
Eggs	0.7	1.5	1.7
Milk ('000 litres fat adjusted)	0.04	0.07	0.07
Beef (dressed carcass)	0.3	0.5	1.2
Sheep (dressed carcass)	0.2	0.4	1.0

GHG emissions from the conversion of pasture land to arable, in overseas production, were calculated based on the extra land required under organic management, multiplied by land use change emission estimates for CO_2 as specified by the British Standards Institute (2011), for a range of countries within and outwith Europe – see Table 7.2 (British Standards Institute, 2011). It was assumed that woodland would not be converted to arable land, as this would release considerably more CO_2 eq than tillage of pasture and would represent a direct conflict with the International Federation of Organic Agriculture Movements (IFOAM) organic principles[14]. GHG emissions and fossil energy use associated with the production

[14] http://www.ifoam.bio/en/organic-landmarks/principles-organic-agriculture

of oilseed rape, wheat and lamb (i.e. commodities imported to the UK from non-European countries) were derived from Pelletier *et al.* (2008) and Webb *et al.* (2013). The environmental burdens associated with crop and livestock products sourced from the rest of the UK and Europe were derived from the Agri-LCA, under the assumption that similar emissions and fossil energy use would occur in these systems (Williams *et al.*, 2006).

Table 7.2: GHG emissions from Land Use Change (from pasture to arable) by region from PAS2050 (British Standards Institute, 2011). These are assumed to continue for 20 years.

Region	**GHG emissions ($tCO_2eq\ ha^{-1}\ yr^{-1}$)**
UK	7.0
Europe	6.5
Other	5.7

As with soil carbon sequestration rates, GHG emissions from land use changes are uncertain but can have a major impact on estimates of agriculture's net contribution to climate change (Smith, 2004). Bearing in mind that not all of the additional overseas land required to compensate for domestic supply shortfalls would necessarily be newly cultivated, a sensitivity analysis was undertaken to explore the effect of different assumptions about the requirements for conversion of land from pasture to arable. Three land-use conversion rates were applied to the additional land requirements as follows:

- Low – where only 25% of the additional overseas arable land required under 100% organic conversion was formerly grassland
- Medium – where 50% of the additional land required was formerly grassland
- High – where all of the additional land required was formerly grassland

The low, medium and high LUC and soil C sequestration estimates were combined in three scenarios, and the results applied to the GHG impact derived from the Agri-LCA models. This approach captures a range of possible outcomes (Table 7.3).

Table 7.3: Soil carbon sequestration and land use change scenarios applied in this study

Scenario	C Sequestration rate	Land Use Change rate
Scenario 1	Low	High
Scenario 2	Medium	Medium
Scenario 3	High	Low

The total GHG emissions arising from overseas conversion of pasture to arable were found to be inversely correlated to domestic organic oilseed rape (OSR) yields, i.e. the lower England and Wales OSR yields were, the greater the demand for overseas land to make up supply shortfalls. Therefore two additional scenarios were applied in the Agri-LCA where: (a) current oilseed rape cultivars were replaced by cultivars which yield more under organic conditions, and (b) organic sunflower imports were used as an alternative to imported oilseed rape. The new cultivar yield estimates were derived from Valantin-Morison and Meynard (2008), and organic sunflower yields from Mazzoncini *et al.* (2006). Overseas land requirements were also strongly related to the assumption, in the Agri-LCA, concerning the amount of overseas land required to produce each tonne of livestock product. The impacts of changing these assumptions were also explored. First, by applying new assumptions regarding the area of land required to support organic monogastric livestock production overseas (Basset-Mens and van der Werf, 2005; Leinonen *et al.*, 2012a, b); second, by adjusting the land areas required for ruminant-meat imports produced overseas in line with the estimates of Wilkinson (2011) and average organic cereal yields for EU and non-EU countries reported in de Ponti *et al.* (2012, see Table 7.1).

The GHG and fossil energy burdens associated with crops and livestock products exported from the UK were also subtracted from the estimate of total environmental burdens, with exported tonnages applied as a fraction of domestic production (data sources in Appendix E). Where production volumes were reduced below the level of domestic demand it was assumed that no exports would occur, i.e. domestic consumption would take priority. In contrast, where the organic scenario resulted in a net increase in domestic production that exceeded rates of consumption, the surplus, once all domestic demand was met, was exported and subtracted from the total GHG and fossil energy use-impact. The estimated GHG and fossil energy burden is therefore that required to meet domestic food demand under conventional and organic scenarios.

7.4 Results

Conversion to 100% organic management, under the organic scenario, caused a considerable drop in crop production compared to the 2010 conventional baseline, in particular for wheat, barley and oilseed rape (Table 7.4). The low output of organic oilseed rape was primarily the result of a much smaller cultivated area due to its relatively low yield, compared to both conventional OSR and organic alternatives. The increase in legume outputs under the organic scenario was a result of an increase in the cultivated area, as required by the rotational constraints in the OLUM. This area would have increased further had the constraint on maximum production area not been reached. The area of potatoes cultivated also substantially increased, and hit the 150% production constraint, primarily due to the usefulness of potatoes in organic arable rotations for weed control. In addition, the high ME output of potatoes per hectare makes this a favoured crop for maximising food ME production. The high ME yield per hectare of sugar beet also contributed to the crop reaching the upper constraint on production area. Grazing livestock numbers increased under the organic scenario, although total carcass outputs did not increase by the same percentage, as a result of lower carcass weights and longer finishing periods under organic livestock management. Monogastric livestock numbers and associated meat volumes fell sharply under the organic scenario, as a result of lower concentrate feed availability and upper stocking rate limits imposed to reflect organic standards.

Table 7.4: Projected crop production volumes, crop areas, livestock outputs and livestock numbers. Livestock meat outputs volumes are adjusted for bone content after Wilkinson *et al.* (2011).

Crops	Conventional 2010 Baseline - '000 tonnes	Organic scenario - '000 tonnes	*Organic as % of conventional*	Conventional 2010 Baseline 000' ha.	Organic scenario 000' ha.	*Organic as % of conventional*
Wheat	13,870	7,145	*52%*	1,815	1,557	*86%*
Barley	3,447	1,513	*44%*	193	218	*113%*
Rye	96	144	*150%*	21	43	*208%*
Oats	537	806	*150%*	62	156	*249%*
Potatoes	3,447	5,171	*150%*	102	239	*234%*
OSR	2,071	152	*7%*	605	4	*1%*
Sugar beet	6,699	5,184	*77%*	118	118	*100%*
Beans and peas	571	857	*150%*	203	229	*113%*
Cabbage	248	170	*69%*	5,759	5,689	*99%*
Carrots	651.2	977	*150%*	9,808	31,796	*324%*
Onions	345	518	*150%*	8,722	16,308	*187%*
Sugar beet	6,699	5,184	*77%*	118,491	118,491	*100%*

Livestock	Conventional 2010 Baseline - '000 tonnes	Organic scenario - '000 tonnes	*Organic as % of conventional*		Conventional 2010 baseline 000' head	Organic scenario 000' head	*Organic as % of conventional*
Sheep meat	184	235	*128%*	Ewes and rams	10,557	16,984	161%
Beef meat	469	509	*108%*	Suckler cows	938	1,579	168%
Pig meat	482	135	*28%*	Sows and boars	427	154	36%
Eggs	504	152	*30%*	Layers	30	10	33%
Chicken meat	716	178	*25%*	Broilers	85	46	55%
Milk (litres x 10^8)	103	65	*63%*	Dairy cows	1,380	1,074	78%

Aggregated GHG emissions and fossil energy use associated with conventional crop and livestock production and under a suite of organic scenarios are presented in Figure 7.1. The land use change and C sequestration scenarios represent sensitivity analyses of rates of land use change, carbon accumulation and oilseed yield. These results illustrate the total impacts from the transition of domestic production in England and Wales to organic, plus 'exported impacts' i.e. production, carbon sequestration and land-use change resulting from the requirement to produce crops overseas to compensate for domestic production shortfalls.

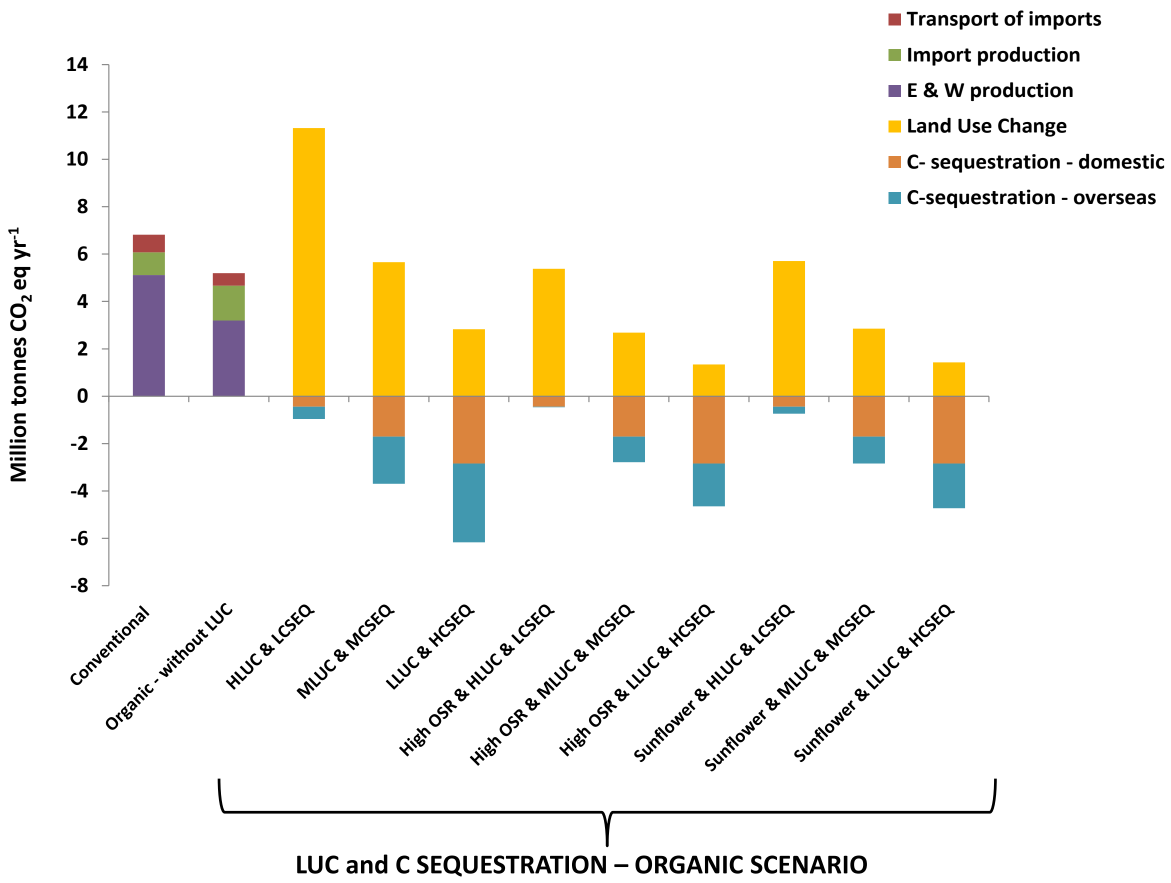

Figure 7.1: Average crop greenhouse gas emissions (GHGs) from all organic scenarios compared to conventional 2010 baseline. HLUC = high land use change, MLUC = medium land use change, LLUC = low land use change. LCSEQ = low C sequestration, MCSEQ = medium C sequestration, HCSEQ = high C sequestration. High OSR and Sunflower refer to high oilseed rape yield and sunflower import scenarios referred to in methods above.

Figure 7.1 shows the production-related GHG savings that could be achieved following conversion to organic methods. The results show that even when the GHGs generated by increased overseas production are accounted for, total GHG emissions for crops still only reach 76% of the non-organic position. GHG emissions associated with transport were also reduced under organic management, as a result of the increase in domestic potato production, and the subsequent reduction in the weight of imports. However, when CO_2 emissions from land-use change are included, the picture becomes less positive. When the rate of land use change is assumed to be high or medium, emissions savings from organic management were more than offset. When land use change rates are assumed to be low and where carbon sequestration for both domestic and overseas organic production is factored in, net GHG emissions are again favourable under organic conversion.

The sensitivity of the results to the oilseed rape yield and the type of oil crop grown is illustrated in Figure 7.2 (high organic oilseed rape yields and sunflower imports result in LUC emissions that were only 47% and 50% of an average oilseed rape scenario, due to a decrease in land-use requirements).

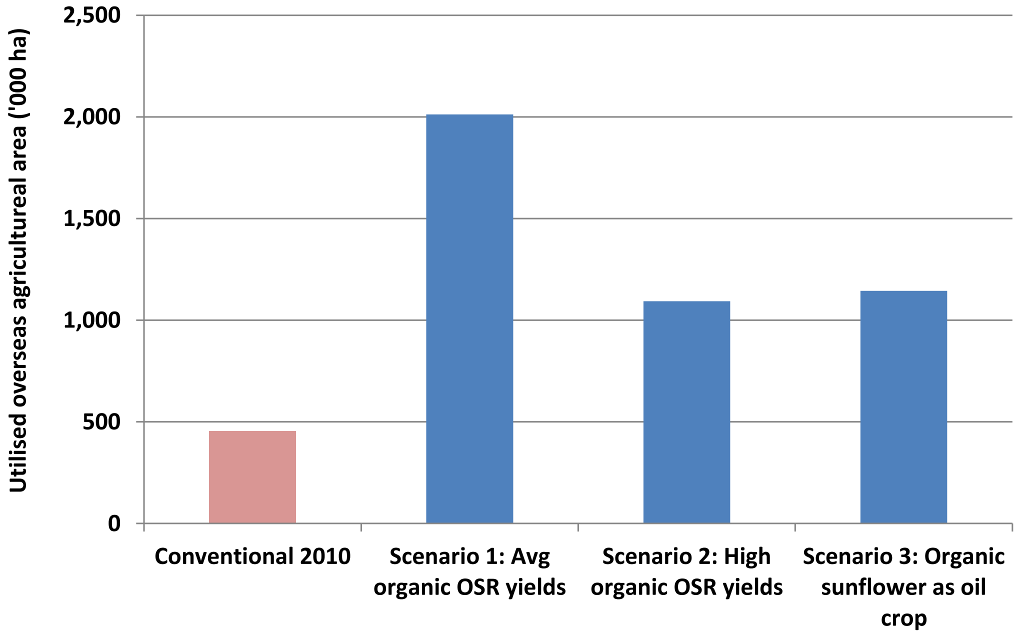

Figure 7.2: Area of land needed for overseas imports under the 2010 non-organic baseline and organic scenarios: crops.

A similar picture is presented for livestock production as a result of lower GHG emissions per unit of product in the organic scenario. For beef and lamb in particular, the production-related GHG emissions were lower under a 100% organic scenario (Figure 7.3). However, when emissions from LUC were included, the emissions savings of the organic scenario were again fully offset (total emissions were approximately 26% higher than the 2010 baseline based on the average of all LUC scenarios in Figure 7.3). When carbon sequestration was included, the total GHG emissions in the organic scenario were still higher at 104% of the 2010 non-organic baseline. Import production constitutes a much greater component of the total GHG footprint under organic monogastric livestock production, as the differences in output were much greater compared to most of the crops, particularly for poultry meat, pork and eggs, due to the large decrease in domestic production (Table 7.4).

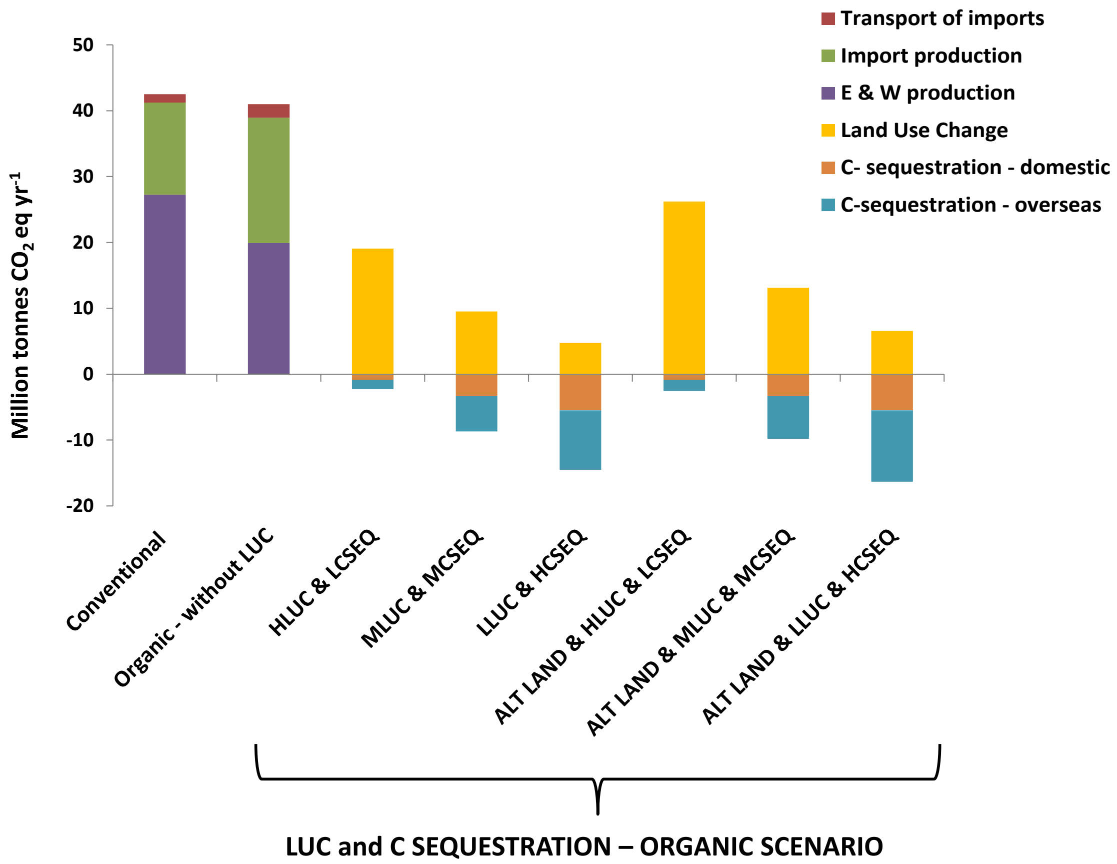

Figure 7.3: Average livestock GHG from all organic scenarios compared to conventional 2010 baseline. HLUC = high land use change, MLUC = medium land use change, LLUC = low land use change. LCSEQ = low C sequestration, MCSEQ = medium C sequestration, HCSEQ = high C sequestration. ALT LAND refers to alternative values for land-occupation associated with livestock production, as described in methods above.

The considerable reduction in emissions for the domestic production element of GHG sources under organic management (i.e. excluding emissions associated with overseas LUC) was a direct result of reduced N_2O emissions, achieved through the avoidance of fertiliser manufacture and lower rates of N application for many organic field crops (Figure 7.4). Emissions were higher per tonne of output for some organic crops, e.g. for organic field beans grown in England and Wales, due to increased rates of N leaching and denitrification under organic management, a result of the increased presence of this crop on wet/heavy soils, although significant volumes of field beans grown for human consumption would have to be exported as a result of low rates of domestic consumption. Cereal crops traditionally receiving lower rates of manufactured N fertiliser under non-organic management (i.e. oats and spring barley) also had greater GHG emissions per tonne under organic management, as the yield reduction offset the savings associated with imported fertiliser in the conventional system. In addition, increased GHG emissions were found for organic crops requiring a much higher fossil fuel input in their cultivation (e.g. organic potatoes and carrots, which require flame weeding/flame haulm removal for weed and disease control).

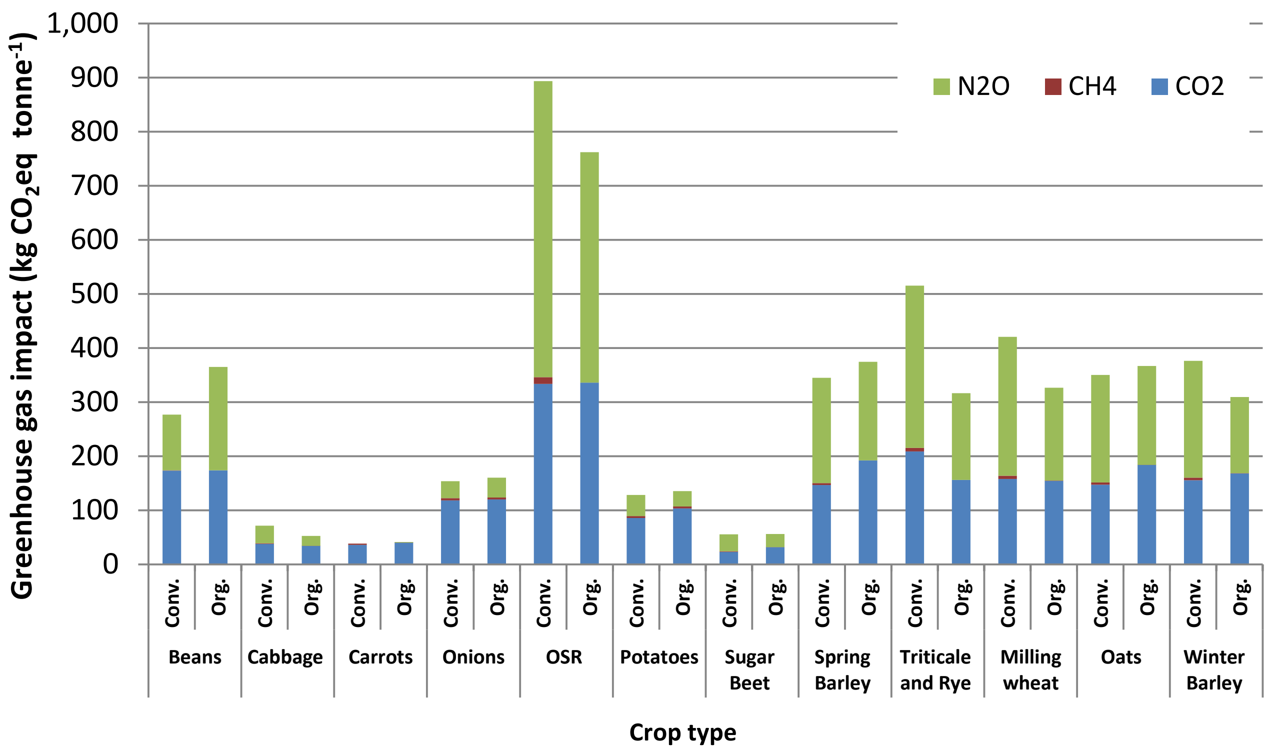

Figure 7.4: Crop GHG emissions per tonne of product (excluding Land Use Change) under organic and conventional scenarios (i.e. adjusted values per tonne from the Cranfield Agri-LCA – the suite of Life Cycle Assessment-based models applied in this study)

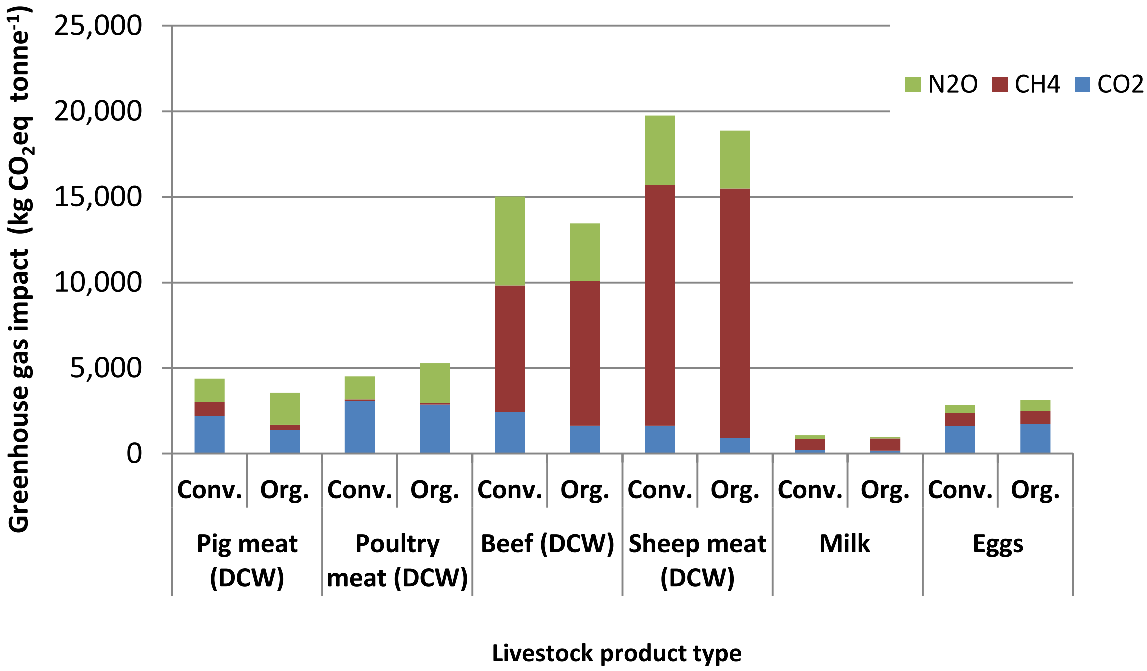

Figure 7.5: Livestock GHG emissions per tonne of product (excluding Land Use Change) by source under organic and conventional scenarios. Meat production impacts are on a Dressed Carcass Weight (DCW) basis

Organic pig production produced lower GHG emissions per tonne of product despite the reduced yield, as the outdoor nature of organic systems greatly reduces requirements for fossil energy used in housing and its associated CO_2 emissions and reduces methane emissions by not having slurry storage, although N_2O emissions were increased as a result of greater leaching and denitrification from manure deposition and application (Figure 7.5). Poultry meat and egg production generated greater emissions under organic management as a result of poorer feed conversion ratios, longer rearing times, higher mortality rates and greater leaching losses compared to conventional free-range and fully housed systems. Organic dairy, beef and sheep production resulted in lower total GHG emissions, although greater forage intake in the organic system increased the total CH_4 contribution per tonne of product.

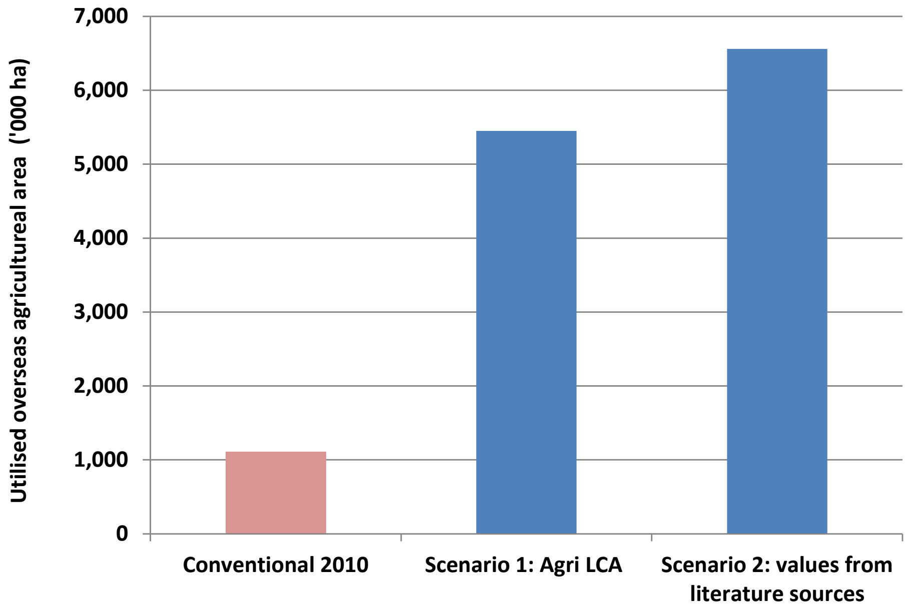

Figure 7.6: Area of land needed for overseas imports under the 2010 non-organic baseline and organic scenarios: livestock.

The requirement for overseas land to make up for shortfalls in domestic supply under the full organic scenario is over five times the amount required in the conventional 2010 baseline when applying the Agri-LCA values (Figure 7.6). This is largely the result of the requirement to increase imports of organic pork, poultry meat and eggs, the overseas land area requirements for which are between 1.8 and 4.1 times the non-organic baseline on a per tonne basis.

When results for livestock, crops and all land-use-changes were combined (Table 7.5) the resource efficiency savings obtained through organic production systems were sufficient to cause a slight decrease in total GHG emissions, despite an increase in transport emissions through increased imports. When emissions associated with the conversion of additional land to arable production were added, there is a net increase in emissions under organic management, although this is offset to a large extent when domestic soil carbon sequestration is factored in.

Table 7.5: Aggregate greenhouse gas emissions (GHGs) under organic scenarios compared to the conventional baseline. LUC = Land Use Change

	Domestic production and imports	Production, imports & Land Use Change (LUC)	Production, imports, LUC, C sequestration
GHGs: organic scenario ($ktCO_2e$)	46,192	63,753	51,313
GHGs: non-organic scenario - 2010 ($ktCO_2e$)	49,312		
Organic GHGs as proportion of non-organic	94%	129%	104%

From a fossil energy use perspective, the organic systems assessed were more efficient per tonne of product for ruminant livestock production and for major field crops such as wheat (see Figure 7.7 and Figure 7.8) or per MJ of food energy produced (Table 6.6). Again this is largely a result of avoiding the use of manufactured N fertiliser, which is derived primarily from natural gas, although as with GHG emissions, some exceptions occurred for crops receiving lower amounts of manufactured N in conventional farming and for organic crops requiring flame weeding. Livestock follow a similar trend to the crop-GHG impacts, with ruminant production systems comparing favourably (in particular sheep meat production systems) as a result of reduced inputs of concentrate feed and associated N fertiliser manufacture. Organic poultry systems were less efficient in terms of fossil energy use as a result of much lower meat outputs per unit of feed.

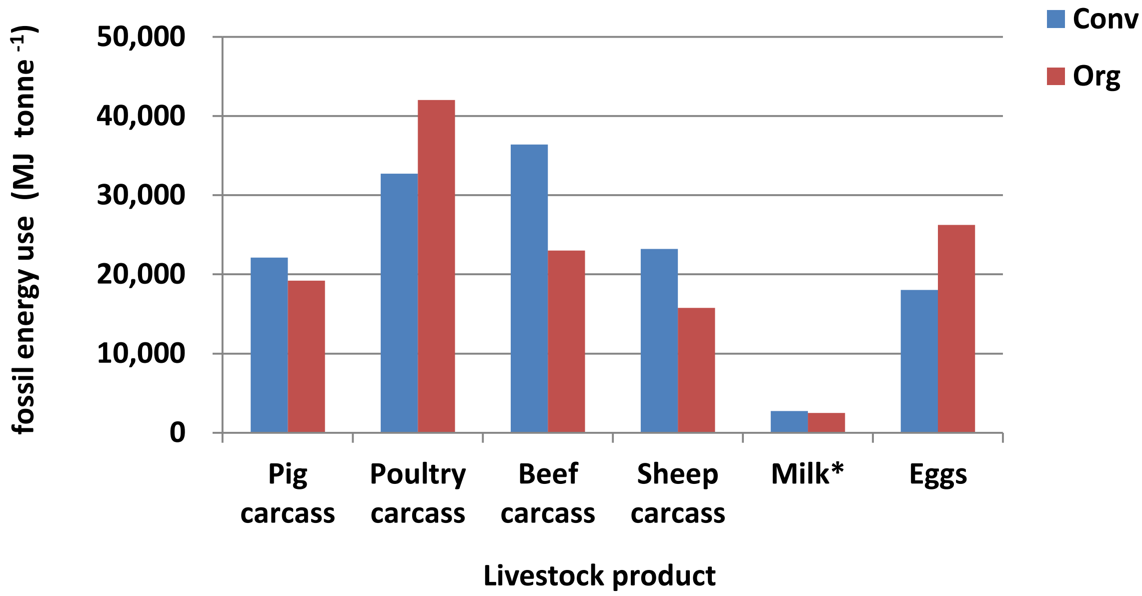

Figure 7.7: Fossil energy use per tonne of animal product under organic and conventional scenarios. * = MJ/'000 litres

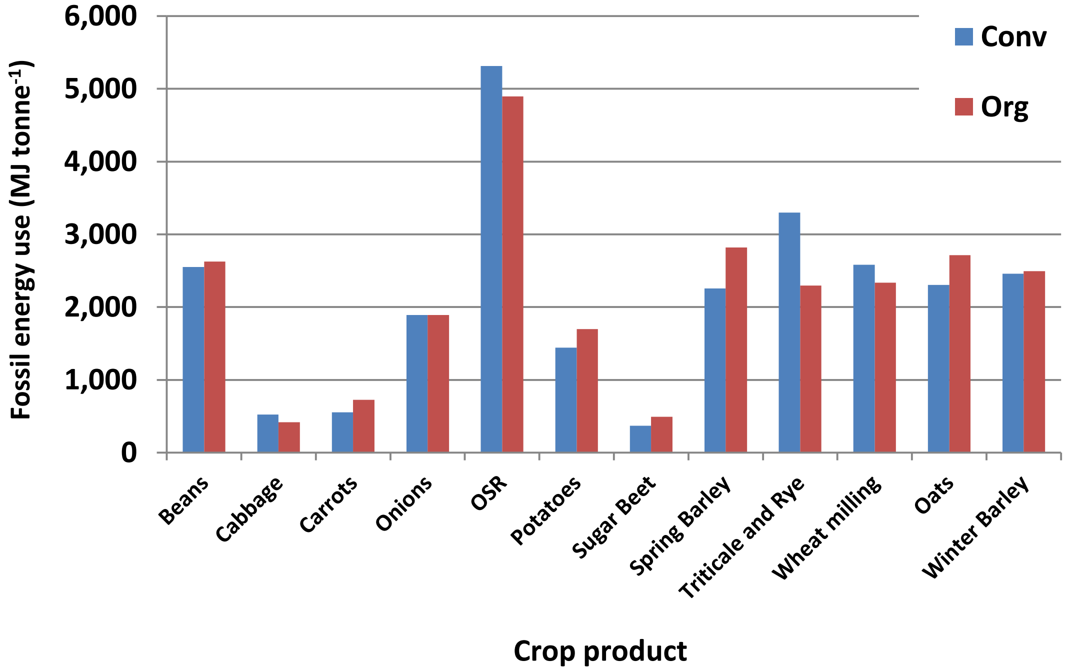

Figure 7.8: Fossil energy use per tonne of crop product under organic and conventional scenarios. OSR = oilseed rape

Table 7.6: Energy ratio (total metabolisable energy (MJ ME) output by crop divided by primary energy used in production, i.e. for beans 4.4 MJ of energy are produced for every 1 MJ of energy in under conventional management. Most primary energy in these cases is non-renewable.

Crops	Energy rato Conventional (OUT/IN)	Energy ratio Organic (OUT/IN)	Difference - organic ratio as % of conventional
Beans	4.4	4.3	97%
Cabbage	2.0	2.5	125%
Carrots	4.1	3.2	76%
Maize silage	2.1	2.2	107%
Oats	4.5	3.8	85%
Onions	1.4	1.4	100%
Oilseed rape	2.3	2.5	109%
Vining Peas	6.0	6.5	109%
Potatoes	1.9	1.7	85%
Spring barley	5.1	4.1	80%
Sugar beet	6.8	5.1	75%
Triticale and Rye	3.5	5.1	144%
Feed wheat	4.3	5.0	117%
Milling wheat	4.6	5.0	111%
Winter Barley	4.6	4.6	99%
Livestock products - adjusted for bone and shell content			
Pig meat	0.2	0.3	115%
Poultry meat	0.2	0.1	78%
Beef meat	0.2	0.3	158%
Sheep meat	0.3	0.5	185%
Milk	1.0	1.1	110%
Eggs	0.3	0.2	69%

7.5 Discussion

Results from the life cycle assessments revealed that whilst substantial gains in the efficiency of non-renewable resource use could be obtained through a large-scale conversion to organic production methods, a major increase in the area of overseas agricultural production would be required, to compensate for production shortfalls if diets were to remain the same. If only 50% of the additional overseas land area were to be converted from pasture, then the GHG savings obtained by the use of low-input methods commonly applied on organic farms would be largely offset. At lower rates of grassland conversion, and when soil carbon sequestration rates associated with organic management are factored in, fully organic production would moderately reduce net GHG emissions from England and Wales agriculture. However, fundamental questions remain, i.e. can the additional overseas land required to make up for production shortfalls be found at all, and if it can be found, can this be obtained solely from the existing base of tilled land? With the demands placed on the existing arable land base continuing to increase as a result of growing populations (Bajzelj *et al.*, 2014), a shift to lower-yielding forms of agriculture, such as organic farming, could be considered infeasible without a fundamental change in consumption habits and/or production systems. The following sections explore whether changes in both of these areas could be made in order to make a full-organic conversion scenario feasible.

7.5.1 Effecting behavioural change towards 'sustainable diets'

It has been highlighted that demand-side measures, including changes to national diets, are essential to ensure progress towards GHG mitigation targets (Bajzelj *et al.*, 2014). Unfortunately, evidence suggests that changing dietary habits on a large scale is likely to be a slow and difficult process. The difficulty results from a range of factors, such as ingrained consumption habits, disagreements between stakeholders on what a sustainable diet consists of, low public awareness of the health and environmental impacts of food choices, consumer purchasing power and a lack of meaningful commercial and/or Government support for change (de Boer *et al.*, 2014; Garnett, 2014b). Although some modest successes have been achieved in changing consumption habits in recent years, (e.g. the Love Food Hate Waste campaign in the UK) the current rate of change in consumption patterns is unlikely to support the land use change needed to yield any significant offsetting of the GHG impacts associated with food and farming even in the medium term (Bajzelj *et al.*, 2014). The development of more conjoined international policy support (for example through the introduction of a carbon tax applied to food purchases and implemented across G8 members) could help to drive change towards lower carbon agriculture in these countries, for example through public procurement and state education programmes. However, more research is required on the efficacy of potential state interventions to drive change in this area and the identification of, and means to overcoming, possible barriers to change, for example conflicts between environmental, socio-economic and nutritional goals (Bajzelj *et al.*, 2014).

Education campaigns to raise levels of food and health consciousness among citizens may help to make the scenario presented here more feasible. Evidence suggests that consumers buying a high proportion of organic food exhibit a greater tendency towards fresh vegetable consumption and vegetarianism (Hamzaoui Essoussi and Zahaf, 2008). This is likely due to the fact that interest in organic food arises out of a more general food and health consciousness and to a lesser extent concern for the environment. A successful public education programme of this kind could therefore yield multiple benefits, i.e. increasing acceptance of less resource-intensive diets, and improving public health (Macdiarmid *et al.*, 2011). The current small-size of the organic sector, and in particular the tendency for organic consumers to be from higher-income households, makes the likelihood of the efficacy of such a public education campaign difficult to predict (Dimitri and Dettmann, 2012). However, with the potential benefits of such an outcome so large, further research on the complementarities between sustainable and healthy consumption, following a widespread public education campaign, must surely be worthwhile.

It may also be possible to encourage food production in urban spaces under a 100% organic scenario, following examples elsewhere and the past. A good exemplar from the UK past would be the Dig for Victory campaign operated during the Second World War. This campaign encouraged all households to take on land, which the family would cultivate, to provide an additional household supply of fruits and vegetables and so address the very pressing national food security problem. From elsewhere in the world other possible exemplars might include Cuba and the Pacific Island states, where horticulture practiced in civic and urban environments make important contributions to ensuring an adequate supply of nutritious food (Altieri *et al.*, 1999).

7.5.2 Improving organic production systems

Closing the yield gap between conventional and organic agriculture would go a long way to improving the feasibility of full organic conversion, especially where major crops, such as cereals, are concerned. Measures to address the N availability limitations and leaching losses commonly occurring with organic management (e.g. making better use of manures and cover crops) are likely to make an important contribution to organic yield improvement strategies (Smith *et al.*, 2016). Improvements in organic rotation design (such as avoidance of ploughing leys in the autumn) and ensuring a more effective and reliable supply of N from biological fixation, may also be required, particularly in the case of stockless organic farms, which can struggle to maintain positive N balances over a rotation (Smith *et al.*, 2016). Allowing sewage sludge application on certified organic land could also help to address commonly experienced phosphorus (P) deficits, which are responsible for yields losses in crops with a high P requirement, e.g. potatoes (Smith *et al.*, 2016).

Replacing crops that perform poorly under organic conditions with similar alternatives could also help to improve the relative performance of the sector, such as substituting oilseed rape with sunflowers. While this crop cannot yet be grown reliably in much of the UK under current climatic conditions, the effects of climate change are likely to encourage a northwards expansion of sunflowers in Europe (Olesen and Bindi, 2002). Establishing more efficient photosynthetic pathways though genetic modification of major crops like wheat, oats, barley and rice could also drastically increase yields under the warmer growing conditions expected as a result of climate change. This would have multiple beneficial effects, such as decreasing land-use requirements and improving water and nitrogen use efficiency (Beatty *et al.*, 2015). Progress in this area has been slow to date, however (Furbank *et al.*, 2015), and international organic standards currently prohibit the use of genetic modification as their use is seen to be in conflict with organic principles and consumer expectation.

With regard to organic livestock production, increasing outputs per unit of land is likely to require a change to international organic standards, for example to relax upper stocking rate limits, which are currently deployed to promote animal welfare and reduce reliance on external inputs. It may also be necessary to revisit exclusion criteria concerning the use of certain manufactured feedstuffs (e.g. synthetic amino acids for poultry). Adjusting standards in this area could also allow for synergies between cropping and livestock dominated systems, i.e. surplus manure produced by more intensive livestock systems could be exported to arable-dominated systems requiring readily available sources of N (Wilkins, 2008). Allowing for greater flexibility in other organic standards, for example with regard to the current requirements for providing animals unfettered access to range, could also help to improve the productivity of organic farming and reduce the associated land requirements. This would be of particular value to the poultry sector, both in terms of overcoming stocking density limitations and as a means to reducing organic poultry mortality rates and disease/predation pressures, as these are elevated in outdoor systems (van de Weerd *et al.*, 2009; Weeks *et al.*, 2016).

Adapting breeding programmes to produce animals better suited to organic farming and standards could also be beneficial. In particular, adapting monogastric breeds for organic management (i.e. developing breeds suited to longer rearing periods and increased reliance on feed foraged from the range and/or using dual purpose breeds) could be considered to reduce the high metabolic pressure created by combining organic diets with breeds more suited to intensive conventional systems, and other factors conflicting with the IFOAM organic principles (e.g. discard of male chicks in egg production systems, van de Weerd *et al.*, 2009).

7.5.3 Organic farming's contribution to GHG mitigation and energy efficiency

The results from this study also highlight the extent to which organic farming practices can lead to improvements in GHG mitigation and fossil-energy efficiency. In particular, for cereal cropping systems and ruminant agriculture, the application of organic farming practices would undoubtedly reduce the reliance on energy-hungry inputs such as manufactured N fertiliser and concentrate feed and save their associated N_2O emissions. The potential benefits that could arise from the wider application of certain organic management practices, such as producing grass swards incorporating clover and other legumes as a means to reducing reliance on artificial nitrogen inputs make such practices worthy of further research. Also worthy of further investigation are the benefits associated with the wider application of mixed farming approaches, commonly applied in organic management, as a means to optimising the efficiencies that can be achieved through a closer integration of crop and livestock operations (Wilkins, 2008). The improvements offered by organic management in the dairy sector, in terms of fossil energy efficiency, also highlights the potential benefits of a reduced reliance on concentrate feeds. However, benefits in the above areas need to be set against the requirements for additional land resulting from reduced yields per hectare or unit of livestock, as observed in this study.

The benefits of soil carbon sequestration for organic systems have also been highlighted by this study. Soil carbon accumulation could offset between 4% and 38% of GHG emissions resulting from food production when accounting for domestic production, imports and CO_2 from overseas land use change. The main routes to increased soil carbon concentrations in organically managed soils are the use of ley/arable rotations and the application of organic manures and composts (Gattinger *et al.*, 2012). Current initiatives promoting lower-input farming approaches may lead to a greater uptake of these practices, which could allow the non-organic sector to capture some of the advantages of organic farming in terms of soil carbon accumulation. It should also be noted that the bulk of any increase in soil carbon concentrations is likely to occur in the early years following organic conversion, and Gattinger *et al.* (2012) found that soil carbon sequestration rates within organic systems were only significantly higher in studies of 20 years or less. In addition, the primary GHG impacts of overseas land use changes are also likely to occur in the first 20 years of conversion from the previous land-use (Williams *et al.*, 2014). In consequence, the net benefits of organic systems, in terms of carbon sequestration could be, over the longer term, close to zero, depending on the amount of carbon sequestered and/or lost following a change in practices.

7.5.4 Methodological critique

There are a number of limitations to the scope of this study that should also be considered. For example the uncertainties that exist in the area of soil carbon sequestration, particularly annual rates of accumulation and saturation points, as well as N_2O emissions can vary greatly depending on local environmental conditions and farm practices. Such uncertainties hinder the development of accurate estimates in both areas (Reay *et al.*, 2012; Smith, 2004). Whilst time constraints prevented the estimation of statistical uncertainties in this study, the results provide a centre-ground estimate, indicative of the nature and general magnitude of the change that might be expected from widespread organic conversion. Although some margin of error in these estimates must be accepted, the estimations can be taken as robust, especially where very large-scale differences between conventional and organic are projected. It is recognised that the focus on GHG emissions and fossil energy use also does not provide a full picture of the extent to which organic farming systems can contribute towards multi-objective and internationally binding sustainability targets (e.g. the Sustainable Development Goals outlined post-Rio+20). For example, the study has not been able to take into account some other benefits that might arise out of a full organic conversion with regard to an increase in on-farm biodiversity or rural development and reductions in pesticide use (Lampkin *et al.*, 2015).

7.6 Conclusions

An environmental assessment of the impacts of a 100% conversion to organic farming in England and Wales revealed that whilst considerable improvements in resource use efficiency could be obtained following a switch to organic production methods, reduced outputs would mean that more imports would be required to maintain food supply. A major expansion in agricultural cultivation overseas, to make up for domestic supply shortfalls, could result, leading to an increase in GHG emissions from the associated land use change. Some or all of this GHG increase resulting from LUC could be avoided if it were possible to increase the commodity supply, for export to the UK market, by means of yield increases on existing organically-managed arable land. The carbon sequestration benefits obtained following a conversion to organic farming could also help to offset emissions increases from the increased production area/intensity, although these benefits would be time-limited and their scale uncertain. Replacing oilseed rape with sunflowers, or another oilseed crop, could also help to reduce the land requirements of organic farming. Some routes to increasing the volume of outputs (productivity) from organic production would require changes in international organic standards. Changes to diet would also help to enhance the feasibility of a large-scale organic conversion.

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CHAPTER 8. DISCUSSION

In this chapter, the results from the work are summarised against the hypotheses set in section 1.2. The implications of the results are also discussed with particular regard to current policies addressing the development of sustainable food systems in the UK. Potential consequences for the organic sector are also discussed from regulatory and farm management perspectives. Alternative approaches and potential areas for future work are considered before overall conclusions are drawn based on the outcomes from preceding chapters.

8.1 Summary of key results in relation to the study hypotheses

8.1.1 Hypothesis 1

A 100% conversion of agriculture in England and Wales to organic practices will not significantly reduce the levels of production for major arable and horticultural crops and livestock products.

The first null hypothesis posed in section 1.2 can be rejected as Chapter 6 reveals that a widespread conversion to organic farming would result in a major reduction in productivity for most crop and livestock products. The production of major cereals (e.g. wheat, barley) would be particularly affected due to lower yields (e.g. organic yields were only 51% of conventional in the case of wheat) and reduced cropping areas as a result of the need for fertility building leys in organic rotations. Outdoor vegetable production would be less affected due to an increased presence of vegetable crops in organic rotations and smaller yield gaps for organic crops of this type. For livestock products, reductions in food production would also be severe, particularly for poultry (meat and eggs) and pork as a result of lower stocking densities and limited feed availability under a 100% organic scenario. Milk production would also be substantially reduced, reaching approximately two-thirds of the production levels under conventional management, as a result of lower stocking rates and reduced milk yields on organic dairy farms. Conversely beef and sheep meat production would increase as a result of increased ruminant livestock numbers on organic farms, resulting from an increased uptake of clover/grass leys, particularly in eastern, cropping-dominated areas of England. Total food outputs reached only 64% of conventional production levels under organic farming when expressed as total metabolisable energy (ME), suggesting that a widespread adoption of organic practices would necessitate a major increase in food imports and/or a substantial change in national diets.

8.1.2 Hypothesis 2

A 100% conversion of agriculture in England and Wales to organic practices will not result in a net increase in greenhouse gas emissions.

The second null-hypothesis can be partly rejected. Although a 100% conversion to organic farming in England and Wales could result in reduced greenhouse gas impacts through savings in non-renewable resource-use and avoided nitrous oxide (N_2O) emissions from manufactured-N fertiliser, savings in these areas are likely to be offset by emissions arising from Land Use Change (LUC), resulting from an overseas expansion of agricultural areas, which may be required to redress domestic supply shortfalls under a 100% organic scenario. Nevertheless, organic farming systems can result in lower greenhouse gas emissions when considering production alone (i.e. without considering emissions from LUC) and the carbon sequestration benefits that organic farming systems provide could help to offset the impacts of an increased agricultural area. The wider adoption of clover/grass leys and N-fixing cover crops could represent an effective and sustainable management option for reducing the greenhouse gas impacts of agriculture in England and Wales.

8.1.3 Hypothesis 3

A 100% conversion of agriculture in England and Wales to organic practices will result in less fossil-fuel use per kg of product

The third hypothesis posed within this study can be partly accepted. Although the low-input nature of organic systems means that most crops and livestock products will be more energy-efficient per unit of product, lower yields / outputs for many organic crops and livestock can lead to worse performance. In particular outdoor vegetable crops requiring flame-treatment within organic systems (e.g. carrots, potatoes) can be less energy efficient due to the high rates of fossil fuel use and lower yields. Ruminant livestock production is often more energy-efficient under organic management due to the energy efficient practice of producing forage in grass-clover leys, whereas monogastric livestock production tends to perform worse as a result of inferior feed conversion and higher mortality rates.

8.1.4 Hypothesis 4

A 100% conversion of agriculture in England and Wales to organic practices will result in less fossil-fuel use per hectare of land

The fourth hypothesis can be accepted, although there are some exceptions (e.g. when the energy content of organic matter/compost used on organic holdings is accounted for) over 80% of the per unit of land area comparisons shown in Chapter 4 revealed lower energy use in organic production through a reduced reliance on manufactured N fertiliser and concentrate feed.

8.2 Key messages for food and agriculture policy

The results indicate that whilst organic farming can reduce greenhouse gas emissions, particularly through improvements in fossil fuel efficiency, the likely expansion in agricultural area that would accompany a widespread adoption of organic methods in England and Wales, to compensate for supply shortfalls, could offset any benefits in this area. Studies comparing the effects of "land-sparing" and "land-sharing" have drawn similar conclusions, i.e. that the environmental impacts from lower yielding forms of agriculture can be greater than more intensive production systems, due to the need for more cultivated land (Green *et al.*, 2005). Although such comparisons can miss 'real-world' effects (in particular the effect of economic drivers on agricultural expansion, Perfecto and Vandermeer, 2010), the conclusions drawn still support the need for more production from fewer resources, as one part of any progress towards 'Sustainable Intensification' (Foresight, 2011). The results from this study suggest that a widespread conversion to organic farming could conflict with such objectives and industry roadmaps promoting greenhouse gas mitigation in England and Wales agriculture (e.g. the Greenhouse Gas Action Plan, Greenhouse Gas Action Plan Partnership, 2012). The results also support the suggestion that recent estimates of the organic/conventional crop-yield gap (e.g. de Ponti *et al.*, 2012; Seufert *et al.*, 2012; Ponisio *et al.*, 2015) are likely to be misleading, as these do not consider the additional land required to support organic production, with particular regard to the fertility-building ley phase within crop rotations (Connor, 2013).

Despite this finding, it is possible that the lower yields obtained within organic systems represent a ‘sustainable optimum’ when considering the right balance between a healthy economy, a healthy population and a healthy environment (Doherty *et al.*, 2017). It is also possible that a shift to typical “organic diets” could help to “manage and not just meet demand” (Ingram, 2017), in particular through lowering meat consumption (higher rates of vegetarianism have been observed in consumers frequently buying organic food products, Hamzaoui Essoussi and Zahaf, 2008) although regular ‘organic consumers’ are a small, self-selecting population and how the whole population would respond to being offered only organically produced food is unknown. The methods commonly used on organic farms could also have greater potential to be successfully woven within agricultural landscapes, compared to separate agricultural intensification and/or land conservation measures. Such an approach challenges the view that environmental protection and food-production are diametrically opposed, instead viewing agricultural systems as part of an integrated whole that can support economically and environmentally viable production systems producing healthy, nutritious food at an affordable price (McIntyre *et al.*, 2008).

With the right adjustments at a societal level, it may therefore be possible to ensure that a widespread adoption of organic production becomes feasible. These adjustments would require a long-term dialogue with the public on the future of the countryside and the importance of dietary change from a range of environmental and human health perspectives. As the main route to market in the UK, retailers could have a key role to play in this context and could be encouraged to take a more active role in the implementation of sustainable food systems, for example though Defra’s 25 year plan for the future of food and farming in the UK (Doherty *et al.*, 2017).

It is also possible that UK policy could support the uptake of “organic practices” within conventional farming (e.g. within integrated farming approaches) to avoid the yield reductions associated with certified-organic farming, whilst obtaining some of the benefits associated with a reduced use of manufactured inputs (e.g. through the use of clover in grassland to replace or reduce the use of manufactured N fertiliser in ruminant livestock farming). Combining the carbon sequestration benefits of organic practices with conventional agriculture could also be considered, particularly in view of the UK Government's commitment to join the '4 per 1000' initiative first proposed at the Conference of the Parties 2016 (Minasny *et al.*, 2017). In particular the mixed farming approaches commonly found on organic farms, which provide greater opportunities for the application of manure on cultivated land, and the use of ley-arable crop rotations, are worthy of further consideration in this context. However, the sequestration benefits that can accrue are time limited and reversible, as well as being highly dependent on the starting position[15].

8.2.1 The importance of context specificity

The results from this study also highlight the importance of farm system type when comparing organic and non-organic production. For example on cereal producing farms in Eastern areas of England, where wheat yields are consistently over 7 t ha^{-1} yr^{-1}, the "opportunity cost" of implementing organic practices is likely to be high, due to the drop in yield and a consequent increase in imports. Conversely in grassland dominated areas, the adoption of organic methods may increase production efficiencies, without substantially increasing land-use requirements (organically managed grass/clover leys can even outyield pure ryegrass swards, depending on the amount of artificial N applied in the conventional system, Lampkin *et al.*, 2005). Developing targeted region-specific support programmes for the development of the sector may therefore be appropriate, as seen in Italy and Spain (Sanders *et al.*, 2011).

The results also highlight the potential benefit of increasing the organic land-areas within the UK field-vegetable sector. The organic yield-gap for these systems is lower than other field crops, such as milling wheat, and increases in uptake may help to encourage fresh vegetable consumption within the UK, as a result of perceived health benefits relating to nutrition and pesticide residue avoidance (Magnusson *et al.*, 2003; Yiridoe *et al.*, 2005; Petersen *et al.*, 2013). Increasing the levels of domestic organic vegetable supply could present difficulties in terms of resource use

[15] In practice, the highest SOC sequestration rate suggested by Gattinger *et al.* (2012) of 0.45 Mg C ha^{-1} yr^{-1} would result in a 9 g C kg^{-1} soil in 20 yr, assuming a bulk density of 1g/cm^3, i.e. approx. 1%, or a typical clayey conventional arable soil with 2% organic carbon increasing to 3% in 20 years

however, with particular regard to manure and land-availability (Chapters 5 and 6). The results from Chapter 4 also highlight that organic glasshouse vegetable, apple and potato production systems are generally less efficient than conventional production in terms of fossil fuel use, as a result of lower yields and similar or higher inputs per hectare. These lower efficiencies could be improved by revisiting the currently high cosmetic standards within the retail sector and/or by developing varieties better suited to low-input systems.

8.3 Key messages for public health policy

Although the focus of this study was on the impact of a 100% conversion on national production, fossil energy use and greenhouse gas emissions, it is possible to draw some tentative conclusions on the extent to which an organic scenario would align with policy drivers for the promotion of public health in the context of national "Eatwell" recommendations and the associated food groups[16] (Macdiarmid *et al.* 2011). These national guidelines recommend increases in starchy carbohydrate, vegetable and fruit consumption and reductions in meat consumption within the UK (Macdiarmid *et al.* 2011). Although the 100% organic scenarios presented in this study align with these recommendations in terms of reducing meat production and availability, the increase in beef and lamb production under a 100% organic scenario could conflict with national recommendations to reduce intake of saturated fats. In addition, the 'missing protein' resulting from reduced meat production under an organic scenario would need to be replaced, although some reductions in this area could be envisaged given the current over-consumption (Hess *et al.* 2015). If diets were to remain the same however, the effect of addressing the shortfall in meat production through increased imports could be disastrous from an environmental standpoint (Chapter 7). Replacing animal protein with alternative sources such as beans, peas, insect meal, algae or sorghum could represent a much healthier and more sustainable option, although consumer acceptance is likely to present a barrier, particularly in view of commercial interests and the cultural significance of meat in society (Boland *et al.*, 2013; Day, 2013; de Boer *et al.*, 2014; Verbeke, 2015). An increase in fish consumption could also help to tackle the "protein gap" whilst

[16] The Eatwell food groups include: 1. Fruits and vegetables, 2. Bread, rice, potatoes and other starch food, 3. Meat, fish, eggs, beans and other non-dairy sources of protein, 4. Milk and dairy products, 5. Food and drink high in fat or sugar (Macdiarmid *et al.* 2011)

addressing recommendations for increased intake of omega 3 fatty acids (Macdiarmid *et al.* 2011), although the potential impact of changing the supply/demand for fish in England and Wales was not addressed in this study. The reduction in starchy carbohydrate production under organic management also represents a conflict with Eatwell recommendations, which state that over 50% of food energy should come from this source, particularly as a reduced availability of these foodstuffs could drive-up costs of staple foods such as bread and rice in England and Wales. In contrast, the increased potato supply should make prices fall, but potatoes need to be cooked in low fat ways to keep them in the "healthy" complex carbohydrate group. The drop in milk production under 100% organic management may also lead negative health effects in terms of the intake of iron, zinc, vitamin B_{12} and calcium. There is also evidence to suggest that iodine concentrations in organic milk are significantly lower, which may lead to adverse effects on neurological development in infants (Bath *et al.* 2012).

Health benefits may also accrue from higher concentrations of antioxidants and lower pesticide residues in food, and improved fatty acid profiles in milk, following a switch to organic production and consumption (Benbrook *et al.*, 2013; Barański *et al.*, 2014). Reduced antibiotic resistance and transfer of resistant bacteria from animals to humans could also result from lower rates of antibiotic use in livestock farming under a 100% organic scenario[17], as recent studies have confirmed lower levels of resistant *E. coli* in organic pigs and a decrease in the prevalence of antibiotic-resistant *Salmonella* in organic poultry (Mie *et al.*, 2017). Whether such benefits would extend to the general population is less clear due to a lack of studies comparing clinical outcomes from the consumption of organic food (Smith-Spangler *et al.* 2016, Tang *et al.* 2017) although organic pork and chicken meat has been found to be less likely to harbour resistant bacteria (Mie *et al.*, 2017). At the same time increased consumption of organic meat in the winter may present a risk factor for *Campylobacter* infection management (Smith-Spangler *et al.* 2012) and reduced antibiotic usage may present animal welfare challenges (e.g. with respect to mastitis control in organic dairy farming, Hovi *et al.* 2003). Nevertheless the extent of the

[17] Although organic standards allow some use of antibiotics, withdrawl periods are longer than those applied within the non-organic sector, and animal products may no-longer be sold as organic following a defined number of treatments (Mie *et al.*, 2017, Österberg *et al.* 2016, Smith-Spangler *et al.* 2012).

challenge faced by increased antibiotic resistance is alarming (over 10 million deaths per year will be attributable antibiotic resistance by 2050 according to a recent estimate, Tang *et al.* 2017) and this could be partly addressed through a broader adoption of organic methods (changes in farming practice would only represent a partial solution, as routine antibiotic use in the human population is unlikely to be greatly affected, Mie *et al.* 2017).

Reduced levels of nitrogen leaching in organic farming could also lead to improved water quality, although organic farming can perform worse in this area when impacts are expressed per kilogram of product (Tuomisto *et al.*, 2012). Improved farmer wellbeing could also result from a widespread conversion, in particular through increased rates of employment, reduced pesticide exposure and improved social cohesion in rural areas (Reganold and Wachter, 2016) although increased labour requirements may pose a challenge as agriculture is currently an unattractive career option in the UK, due to a general perception of labour intensive working conditions and low pay (National Centre for Universities and Business, 2015). The UK's exit from the EU is also likely to create migrant labour shortages, an issue which is likely to be particularly relevant for labour intensive areas such as horticulture (Grant *et al.* 2016, Sumption, 2017).

Looking closely at how areas such as public health and agricultural subsidy can be integrated successfully could be an important part of ensuring a successful implementation of 'organic methods' in food and farming systems. There has also been considerable discussion on the role for education and public information campaigns in creating economic conditions conducive to sustainable and healthy consumption, although it seems likely that there are no quick fixes in this area and a multi-stranded, long-term approach is likely to be required to effect change.

8.4 Key messages for the organic sector

The results presented in this thesis can also inform the continuing development of the organic sector, by highlighting some of the challenges that could be faced following an expansion of organic production. In line with previous studies (e.g. Seufert *et al.*, 2012), a common theme has been lower productivity within organic rotations, as a result of lower nitrogen availability. Results from Chapter 6 illustrate that improving the supply of nitrogen within organic rotations could help to improve the productivity of organic farming across a range of system types, in particular for 'staple' crops such as wheat and sugar beet, which reached 50% and 80% of the non-organic baseline under a 'high-N-fixation' scenario. Crop breeding programmes could also help to improve yields through the development of varieties better suited to low-input systems (over 95% of organic crop production currently relies on varieties bred for high-input conventional systems, van Beuren *et al.* 2011). Ongoing research within the EU Horizon 2020 Programme could help to address this issue, for example through work taking place within the DIVERSify[18] project which aims to identify optimal crop species mixtures as a means to improve yield stability, reduce pest and disease damage, and enhance stress resilience in organic and low-input agriculture.

Results from the NDICEA modelling also illustrate the importance of effective clover-ley establishment in maintaining an adequate supply of N over an organic rotation (Chapter 5) with poor performance in this area leading to negative N balances over a rotation, particularly within the stockless organic systems which do not have a 'back-up' N supply in the form of livestock manure (Smith *et al.*, 2016). Improving the timing of ley establishment and cultivation would also help to improve the productivity of organic systems, and serve to address some of the challenges associated with the synchronisation of N availability and uptake in organic farming (Chapter 5 and Torstensson *et al.*, 2006).

The area of land that must be devoted to fertility building under organic management also presents a significant challenge when comparing total productivity within organic and non-organic agriculture. Studies that have assessed the organic/non-organic yield gap have missed this important issue by focussing on individual crops, rather than farming systems (Connor, 2013), although lower outputs from organic farms were confirmed by Lampkin *et al.* (2015), who compared the net-productivity of organic and non-organic farms within the UK Farm Business Survey (FBS). Chapter 6 illustrates that reducing fertility building-ley areas within organic rotations could help to improve the productivity of organic systems at a national level, although the wider adoption of such intensive approaches may lead to soil-N deficits, as illustrated

[18] Designing InnoVative plant teams for Ecosystem Resilience and agricultural Sustainability. Project website: https://www.plant-teams.eu/

within the ‘stockless’ examples in Chapter 5. Pests, diseases and weeds may also increase, as the ley represents the primary method of controlling these factors (Lampkin, 2002), although herbicide resistance has led to some weeds becoming a serious issue in non-organic production, leading conventional farmers to adopt organic practices (e.g. the use of ley periods for blackgrass control, *Alopecurus myosuroides*, Moss and Lutman, 2013).

Intensifying livestock management could also improve the productivity of organic farming. At higher stocking rates, outputs from organic production increased considerably (e.g. milk production reached 90% of the non-organic baseline within a “high-stocking rate” scenario – Chapter 6). Intensifying poultry production also improved the performance of the sector in terms of total food output, although the production gap between organic and conventional for this livestock sector remained relatively large. Switching or combining modern broiler and layer breeds with geese farming on organic land may also improve the productivity and resource use efficiency of the sector, in particular as geese are able to graze grass-ley areas and can assist with weed control in some organic field crops such as potatoes (Hermansen *et al.* 2002).

Further improvements in the productivity of organic poultry are likely to require a change in international organic standards, in particular to allow for increased bird numbers per hectare and the use of synthetic amino acids in feed, the prohibition of which can negatively affect feed conversion rates, mortality and N excretion (van de Weerd *et al.*, 2009; Dekker *et al.*, 2012; Steenfeldt and Hammershøj, 2015). Results from the energy efficiency study have also served to highlight the problems encountered when combining low-input modes of farming with high genetic potentials within the poultry sector, underlining the importance of focusing organic breeding and/or system redesign efforts on this problematic area. These and other issues relating to the sustainability of the organic livestock sector are discussed in Appendix A.

Increased manure availability from more intensive livestock production could help to make organic systems more productive, particularly if this measure were combined with reduced areas of fertility building ley in organic rotations. Chapter 6 illustrates that higher stocking rates on organic farms could improve the productivity of both crops and livestock at a national level, whereas at lower stocking rates, the net production of N-hungry crops such as potatoes and sugar beet reduced considerably. Allowing for greater use of imported manure from other countries could also help to improve food outputs under an organic scenario, although the economics and disease risks associated with transporting manures and slurries over long distances may be prohibitive. Such approaches could also result in decreased organic N stocks in the country exporting the manure, and may therefore be unsustainable in the longer term. Chapter 5 also highlights the potential benefits that sewage sludge application could provide in terms of improving P and carbon balances within organic rotations, and the current prohibition of sewage sludge application within international organic standards seems to represent an outdated and unjustified directive, that is unnecessarily penalising organic producers and potentially affecting the long-term sustainability of the sector (Smith *et al.*, 2016).

With the advent of the “Organic 3.0” initiative and a new EC organic regulation there may be opportunities to revisit organic standards with regard to contentious areas such as the prohibition on synthetic amino acids and sewage sludge (Arbenz *et al.*, 2016). Although the latest proposal for the new EC regulation was poorly received due to a lack of detail and clarity in its specifications, it is possible that a provision to supplement or amend its elements through ‘delegated acts’ may allow for greater flexibility in the use of imports, although these provisions may be double-edged if the Commission seeks to make amendments without adequate stakeholder consultation (Padel and Woodward, 2014). The new organic regulation is expected to come into force in 2018 or 2019, following a period of extensive consultation with IFOAM and its members.

From a developmental perspective, the work presented also raises a question on the future direction of the organic sector, specifically what the organic sector is ‘for’ in the 21st Century and whether the current organic standards are fit for purpose. In this context it is pertinent to recognise that the early proponents of organic farming (e.g. Albert Howard, Lady Eve Balfour, Rachel Carson) suggested an alternative mode of agriculture as a result of the perceived risks and observed damage to ecosystems resulting from the use of manufactured fertiliser and pesticides between 1915 and the early 1960s. Whilst conventional farming has moved on considerably since this period, both in terms the technology and knowledge applied, it could be argued that the organic sector has not adapted to the same extent. This lack of development has led to some unintended outcomes, for example with regard to the use of 'permitted inputs' in organic farming (e.g. copper based fungicides which are still used in modern organic systems, despite the considerable human and ecosystem health risks, and the fact that safer products have been available for some time, Edwards-Jones and Howells, 2001; Trewavas, 2001). If the overarching aim of organic farming is to create "integrated, humane, environmentally and economically sustainable production systems" (Lampkin *et al.*, 2015) it could be argued that such outdated rules and regulations compromise the core objectives of the sector by ignoring latest developments in order to maintain a mode of agriculture that is considered more 'natural' (Trewavas, 2004). Such outdated and inappropriate standards are also likely to contribute to the yield gap in organic farming systems, whereas a more flexible approach, i.e. one that takes into account new technological developments and the radically changed conventional alternative, could help to address some of the key challenges that are faced by organic producers with particular regard to soil health, N availability and pest and disease incidence (Trewavas, 2004). It may also be possible to envisage a "graded" or "ranked" approach to organic certification, where some of the currently-prohibited inputs are permitted at lower-levels of compliance. Such an approach could allow for substantial improvements in the environmental efficiency of farming, capturing the 'best of both worlds' through a combined approach of organic production methods and application of the best available technology

At the same time the maintenance of soil fertility, closed nutrient cycling and other core principles of organic farming are currently under-represented in organic regulations which instead focus on the distinction between 'natural' versus 'manufactured' inputs, a development that can be traced to the purchasing motivations of 'organic consumers' (Seufert *et al.*, 2017). This reductionist approach to standard setting may lead to organic farming being defined by the lowest common denominator, i.e. as a system that meets the burgeoning consumer demand through input substitution rather than sustainable system design (Seufert *et al.*, 2017). If organic farming is to play a useful role in the development of sustainable agricultural systems it will be necessary to think again about the sector's aims (i.e. whether organic farming is about meeting consumer demand for pesticide-free food or the development of a holistic and integrated approach to improving soil, ecosystem and human health, Seufert et al., 2017). Once these aims are clarified, the sector could ensure that they are more adequately met through a consistent approach to international standards, and thereby help increase consumer confidence and producer uptake (Seufert *et al.*, 2017).

8.4.1 How likely is a 100% organic scenario?

Although the study presented suggests that the impacts of a 100% organic conversion on food supply could be severe, it is necessary to consider how likely such an extreme scenario really is within a modern food systems context. Although the current small size of the sector would suggest that organic farming is likely to remain a niche area for the foreseeable future (organic farmland makes up less than 1% of the global agricultural area, Willer and Lernoud, 2016) organic represents one of the fastest growing food sectors (Seufert *et al.*, 2017) although the senstivity of the market to prevailing economic conditions was illustrated by the sharp contraction in UK organic land following the economic crisis in 2008/9, which affected consumers' willingness to pay more for food (Soil Association, 2012).

Despite such sensitivities, an exploration of the barriers to and drivers for a 100% conversion warrants further consideration, in particular to determine how further expansion of organic farming could be encouraged in line with the aims of national action plans (e.g. Scotland's 'Organic Ambitions'[19] and France's action plan for Agroecology[20]). With this in mind, an international panel of 27 experts from EU government departments, agri-business, research and organic advocacy organisations gathered in Paris, France in May 2017 to discuss the drivers, barriers and impacts related to a 100% uptake of organic farming. The group also considered the likely impacts of the scenario from a food security and long-term sustainability perspective. The results from this discussion are summarised in Table 8.1.

[19] http://www.gov.scot/Publications/2016/01/4353

[20] https://agroecology-appg.org/ourwork/presentation-on-the-french-agroecology-action-plan/

Table 8.1: Summary of key drivers, barriers and impacts associated with a '100% organic' conversion scenario. Source: expert workshop held at INRA Paris in May 2017.

Drivers	Barriers	Impacts
Food scandals (e.g. 'horsegate' in the UK)	Resistance to change in 'conventional' agriculture	More vegetables but less meat overall
Investment in R&D for development of organic systems	Use of GMOs in conventional farming leading to 'contamination' of organic crops / livestock	More agricultural land (land expansion)
CAP reform encouraging greater diversity in crop rotations via agricultural subsidies	Organic / conventional yield gap	Less unemployment, in particular through 'new-actors' within the value chain
Easier certification	Uncertainty – particularly in organic market and future climate	Increased yield variability
Environmental / energy crises	Lack of a focus on a wide range of ecosystem services in agricultural research	Increase in farmer revenue and profitability
Increased profitability following conversion process	Costs of certification	More 'closed' food production systems
Perceived health benefits (e.g. pesticide residue avoidance, improved antioxidant content in foods)	Increased labour requirements on organic farms	Increased arduous manual labour on farms Reduced human health costs (individual and collective)

The results from the discussion suggest that whilst considerable drivers currently exist for the continued growth of the sector, considerable barriers are likely to limit the extent to which organic agriculture can become prevalent within an individual country or region. In particular the costs of conversion, uncertainty over organic yields and the development of the organic market and resistance to uptake were listed as primary limiting factors. Although policy support (e.g. via the CAP and country-specific action plans) may help to overcome these issues, the present uncertainty over the impacts of a large-scale conversion, combined with social and economic barriers, present considerable challenges to the expansion of the sector.

In addition the driver of increased profitability may be an illusion created by the current niche status of the sector - if organic farming becomes a ‘new-norm’ then any advantage or disadvantage in terms of price premia is likely to be eradicated. The impact of reduced human health costs is also under-supported in scientific literature. Although some recent studies have found improved antioxidant contents and lower pesticide residues in organic food (Benbrook *et al.*, 2013; Barański *et al.*, 2014) others have found a lack of evidence for any nutritional benefit (Smith-Spangler *et al.* 2012). The expert discussions also overlooked the increase in on-farm pests and diseases that may result from a widespread conversion to organic farming. In this context it is important to note that most organic units within the UK operate within a ‘sea’ of conventional farming, i.e. farms where pests and diseases are controlled through the use of pesticides (Hole *et al.* 2005). Without this preventative ‘buffer’ in place it is possible that the pest and disease burden resulting from a 100% conversion to organic farming may increase to the point that current organic yields become unsustainable. Conversely, it may be possible that a large-scale uptake of organic farming provides sufficient biological diversity for natural methods of pest and disease control to become established at a wider scale and ‘self-regulate’ disease and pest burdens associated with a widespread uptake of organic practices. The current small size of the UK organic sector prevents an empirical assessment of such effects, however it may be possible to investigate the influence of a wider-uptake in some other European countries (e.g. in Austria and Sweden, where organic land makes up 19% and 16% of the total agricultural area, Willer and Lernoud, 2016). It should also be noted that whilst on-farm employment within an individual country would be likely to increase under a 100% organic scenario, as a result of increased labour requirements, this could be offset by a reduction in employment within the food processing sector, due to an increased reliance on food imports processed overseas.

It is interesting to note that the impacts predicted by the group (Table 8.1) are broadly in line with the results described in in Chapters 4, 6 and 7 of this study, concerning the’ human labour cost, increase in agricultural land areas and the overall effect on meat and vegetable production. The broader costs and benefits highlighted through this exercise (e.g. relating to human health and the development of more closed production systems) also serve to highlight the somewhat limited assessment approach.

8.5 Study limitations and alternative approaches

This approach taken within this study, while more detailed and nuanced than much that has gone before, was inevitably somewhat limited by resources, time and data availability. As a result, the methods included some assumptions that were necessary given the scale of the study and the presence of 'unknown quantities'. The latter issue particularly affected the choice of objective function within the linear programming model (i.e. maximisation of food production as metabolisable energy – ME) as it was considered to be infeasible to predict the economic conditions that would prevail under a 100% organic conversion. The linear programming approach could therefore be considered unrealistic, as it does not adequately represent the business goals of farmers. An alternative approach could follow a more 'traditional' route, i.e. applying economic variables, constraints and parameters specific to the organic sector to maximise the profitability of UK agriculture, under organic constraints (e.g. concerning stocking rate or necessary crop rotations).

Financial data on the UK organic sector are readily available from a range of sources (for example within the organic sample of the Farm Business Survey, and the Organic Farm Management Handbook (Lampkin *et al.*, 2014) and could be applied in such a development although due to uncertainties over the price that would be paid for agricultural commodities in a 100% organic scenario, a considerable range of values would need to be explored. Alternatively the optimisation modelling approach could be adapted to minimise the difference between observed outputs of agricultural commodities within a given region, and the outputs from the same region under organic conditions (an approach suggested in Jones and Crane (2009). Whilst this approach could allow for greater flexibility in terms of the area of land allocated to each farm type within a given region, it may present an unrealistic scenario given the tendency of non-organic farmers to maintain the same or a similar enterprise mix post conversion (Howlett *et al.*, 2002). Constraints concerning land-suitability would also need to be introduced to avoid erroneous allocation of land-uses (e.g. arable farming in the uplands).

A much simpler approach would entail the application of a static, spreadsheet-based model representing typical organic farms across the available land-base within individual regions. Capturing the effect of soil and rainfall class and/or the transfer of manure and livestock feed between farms could present difficulties within such an approach. In addition the model would be dependent on user-assumptions concerning the intensity of production in a given region with regard to stocking rates and crop area (although this could be informed by regional level data from the FBS, the model would be unable to vary the levels of production endogenously). Alternatively current levels of non-organic production could be adjusted manually, based on the difference between conventional and NDICEA-adjusted organic yields. This would build on one of the approaches applied by Jones and Crane (2009) which relied on FBS data to estimate organic/conventional yield differentials and applied these as ratios to the national levels of production for key crops and livestock products. As mentioned in Chapter 6 the FBS is known to over-represent larger, more commercial farms. Using more representative organic yields could improve the accuracy of the productivity estimates presented in Jones and Crane (2009), although this approach would also miss farm trade-interactions (e.g. with regard to livestock manure and feed).

It should also be noted that whilst the approach taken in this study was somewhat limited, it does provide an updated and improved estimate of the production and environmental impacts associated with a 100% conversion to organic farming in England and Wales. In particular through the biophysical constraints implemented within the OLUM, the assessment approach allows for the limited supply of N, livestock feed and agricultural land across a range of organic farming systems. The greenhouse gas assessment also allows for a more considered and detailed overview of the impacts that could be incurred following an adoption of organic management across England and Wales, and includes conservative estimates of the soil carbon accumulation and land use change that could occur under a range of scenarios. The discussion of the results derived within this thesis also provides an objective and balanced overview of the benefits and dis-benefits of the organic approach and useful recommendations on areas where the sector could improve current practice.

8.6 Recommendations for future work

The work presented has also highlighted some potential areas for future work, with regard to the modelling approach and the data that has been derived. Potential developments are briefly outlined in the following section.

8.6.1 Developments to the OLUM

A constructive and useful development of the OLUM could entail an adjustment of the model's structure and data sources, to allow for its application in other countries, or at a global scale. The latter approach could involve the substitution of UK regions with broader geographies (e.g. Western Europe, Africa, and Central Asia). Whilst this would considerably increase the model's size and data requirements, it would help to provide a more accurate assessment of the impacts of a 100% conversion by covering areas where non-organic crop yields and livestock productivities are relatively low compared to the UK. It has been suggested that the application of organic farming methods may actually increase yields and reduce net crop losses in such areas (e.g. in Sub-Saharan Africa) in particular by improving soil health and resilience to the effects of climate change (Auerbach *et al.*, 2013).

Fundamental changes in the OLUM structure could also be applied to explore the effect of optimising public goods or ecosystem service delivery from agriculture under constraints of a minimum food supply, rather than the current approach of maximising food output. This could involve a conversion of ecosystem services to monetary values, building on the work of Chatterton *et al.* (2015) who quantified the net-ecosystem value of the UK livestock sector by attributing monetary costs and benefits to a range of 'provisioning', 'regulating', 'cultural' and 'supporting' services (i.e. categories used to classify ecosystem service provision within the Millennium Ecosystem Service Assessment).

8.6.2 Developments to the NDICEA-based yield adjustments

The yield modelling approaches could also be developed to incorporate a wider range of factors. Within this study only the effect of soil type and rainfall conditions were captured, as both can influence N availability within organic systems, and are therefore highly likely to affect crop-yield. Other factors such as pest and disease incidence, weed burdens and soil compaction are also likely to affect productivity in organic and non-organic systems (Lampkin, 2002). In addition some soil types may be better suited to certain crops than others (e.g. carrots and other root crops are likely to be easier to establish on lighter soils). These and other factors could be incorporated within an updated version of the NDICEA model, or a new model tailored to organic systems.

Despite its limitations, Chapter 5 demonstrates that the NDICEA model represents a very useful learning framework for organic farmers, and its performance on a range of sites across the UK is an encouraging sign. The further development and application of the model could therefore be encouraged, to help organic farmers operating in the UK optimise performance with respect to nitrogen-use efficiency and P and K balances. A useful development could also involve the incorporation of two of the modelling approaches used within this study (i.e. dynamic nitrogen modelling and static land-use modelling). Introducing a temporal component to the OLUM could be required as part of this. This could be achieved by adding the algorithms used within NDICEA to the OLUM, in addition to data sources on local environmental criteria (e.g. daily rainfall, temperature and evapotranspiration).

8.6.3 Developments to the scope of the assessment

The spatial resolution of the assessment approach could also be improved to allow for calculations of production over a 5km^2 or 1km^2 grid, potentially resulting in greater accuracy in the land-use/food output and environmental impact predictions. A simpler and less labour intensive adaption could involve adding other plant and livestock nutrient requirements to the nutrient balance calculations (e.g. with regard to P and K supply/demand per tonne of product) within the OLUM to explore the extent to which over/under supply of these nutrients could influence total productivity and environmental performance. Similarly, results from LCA-based greenhouse gas and fossil energy use assessments could be incorporated directly within the OLUM (i.e. as kg CO_2 eq per tonne of product) in order to explore the effect of alternative scenarios such as “minimisation of greenhouse gas emissions” or “maximisation of fossil energy efficiency” to further explore trade-offs between production and environmental impact.

Alternatively, the OLUM’s objective function could be adapted to focus on the provision of safe, healthy food, building on recommendations within recent projects on the contribution of defined food groups to the UK's national diet (Macdiarmid *et al.*, 2011). Expanding the OLUM to consider the effect of changes in the food system in England and Wales could also reveal useful insights, e.g. the extent to which a reduction in food waste could make a 100% organic conversion scenario more feasible. This approach could allow for an exploration of tensions between health and sustainable production, in particular through an assessment of the extent to which changes in the agricultural industry would influence national-food consumption and public health, or vice versa. The latter would make the model more economic in its focus, and other models may be better suited to assessment of this kind (e.g. IMPACCT and CAPRI - also developed in GAMS – see Chapter 3).

The greenhouse gas and fossil energy-use assessments presented in Chapter 7 could also be expanded to consider a broader range of environmental criteria such as eutrophication, pesticide and water use. The Cranfield Agri-LCA provides a basis for such assessments, and additional chapters / papers could apply the same data sources used within this study to explore the relative impact of organic agriculture using these additional assessment criteria. The biodiversity impact associated with the change in land-use following a switch to organic production methods could also be assessed in an LCA framework through an application of characterisation factors developed by Knudsen *et al.* (2017).

The approach to assessing soil carbon sequestration within this study is also fairly rudimentary, i.e. "flat rates" were applied to the domestic and overseas land areas based on the sequestration rates reported in Gattinger *et al.* (2012). A more detailed assessment approach could consider the C-stock changes that would occur following the widespread implementation of organic rotations and cropping systems through an application of biophysical models (e.g. Roth-C which has been calibrated for a range of agri-climatic zones, Coleman and Jenkinson, 1996). Future work could also consider whether the carbon sequestration benefits obtained through the implementation of organic practices could be allocated to individual crops and/or livestock products, and therefore expressed on a 'kilogram of product' basis. Others have achieved this by considering the addition of carbon from manure and above/below-ground crop biomass within a range of organic rotations (e.g. Knudsen *et al.*, 2014). The lack of a consistent framework for the allocation of soil-carbon stock changes in product carbon footprints using an LCA framework hinders developments in this direction (Goglio *et al.*, 2015).

CONCLUSIONS

The aims and objectives of this research were to investigate the extent to which a widespread conversion to organic farming would meet demands for a more productive yet lower environmental impact agriculture in England and Wales. The results revealed that whilst organic agriculture can offer some improvements in resource-use efficiency, a widespread conversion would result in a major drop in productivity, in particular as a result of limited N availability, lower stocking densities and a major increase in the land area devoted to fertility building ley crops. In addition, whilst fossil energy use is likely to be reduced within a 100% organic scenario, the greenhouse gas benefits obtained through a reduced use of manufactured inputs would be offset by an increase in cultivated land-areas, and the associated emissions from land-use change.

At the same time, the results reveal that considerable improvements in fossil energy and greenhouse gas efficiency are possible within some farming systems where yield reductions are lessened, i.e. within ruminant agriculture and some outdoor cropping systems, although the differences between the systems are often marginal (organic agriculture does not currently offer a radical alternative to most systems operating in the UK). The results also suggest that reducing the area of fertility building ley within organic rotations could considerably improve the outputs from organic farming, although positive N balances in such intensive rotations may be difficult to maintain (Chapter 5). Revisions to organic standards with particular regard to the use of permitted inputs may also help to address lower productivity and environmental efficiencies within some particularly problematic areas (e.g. organic poultry production, Chapter 7).

From a nutrient management perspective, the results from this study conflict with generic claims that organic farming can foster improvements in soil quality as many of the crop rotations assessed in Chapter 5 were deficient in P and K and the stockless organic rotations struggled to maintain a positive N balance, particularly on wetter and lighter soils. In particular, the results from Chapter 5 suggest that organic cropping systems are inherently more vulnerable to poor ley and/or cover crop establishment, due to a reliance on these sources of fertility which release nutrients in response to environmental conditions rather than crop demand. The deficits of phosphorus observed in Chapter 5 could be partly addressed by allowing sewage sludge application on certified land although this would require a change in international organic standards, whilst potassium deficits could be offset by weathering of mineral reserves, depending on the site and management conditions. The energy efficiency study (Chapter 4) also highlights that any claims that organic farming is more energy efficient than conventional should make note of some important exceptions by crop and livestock type with particular regard to monogastric livestock, although the same study found that the commonly held view that organic farmers' require more fossil-fuel for crop cultivation may not be justified.

The system-level modelling approaches applied within this study have demonstrated the value of applying a range of assessment methods to better-understand the trade-offs associated with contrasting food production scenarios and the importance of functional units in determining the relative impacts of organic and non-organic farming. The results from this study also concur with recommendations to take a food-systems perspective, to understand the full-impacts of contrasting land-management practices and diets, through consideration of broader supply and demand aspects (e.g. the extent of any complementarity between organic production, food waste reduction and healthy consumption habits) although the consideration of these broader elements was beyond the scope of this study.

In summary the results from this study suggest that generic claims that organic agricultural systems have an intrinsic potential to reduce GHG emissions are misguided, as they do not consider (a) the expansion in agricultural area that a widespread adoption of organic farming entails, or (b) reductions in resource-use efficiency for some important crops and livestock products and (c) limits to soil carbon accumulation over time. Despite this some organic systems can clearly offer improvements in resource use efficiency, in particular through lower fossil energy requirements, although integrated approaches, incorporating the best of organic and conventional production methods, could offer optimal solutions for the development of environmentally sustainable food and farming.

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Appendix A Book chapter: Can conversion to organic methods contribute to GHG mitigation and improved energy efficiency in livestock production?

A.1 Abstract

Central objectives of organic farming are reduced external inputs and increased long-term sustainability. Whilst there may be local improvements in resource efficiency, the extent to which organic livestock systems contribute to net greenhouse gas mitigation and fossil energy efficiency gains depends on the livestock type, the unit of comparison, and the system boundaries in time and space. This chapter explores these questions and gives examples where organic systems could be improved. It considers soil carbon sequestration in organic livestock systems. Individual cases are used to show how organic producers are implementing measures to improve environmental efficiency in practice. Challenges and opportunities for the sector are discussed from research, farm-practice and regulatory perspectives.

A.2 Introduction

Livestock's contribution to global warming accounts for about 18% of global greenhouse gas emissions, and over 80% of agricultural land is currently used for livestock production (Smith *et al.*, 2013; Steinfeld *et al.*, 2006). Consequently, livestock systems represent the main component of agriculture's global warming impact (Idel, 2013) although the relative contributions vary considerably by species and production system, with ruminant livestock particularly significant, as a result of the global warming potential of methane (CH_4) from enteric fermentation (see Figure A.1), which is now considered to be at least 28 times greater than CO_2 over 100 years (Trottier, 2015). With over 25 million domestic ruminants added to the planet each year over the past 50 years, CH_4 emissions from livestock production are increasing and ruminants account for about a quarter of anthropogenic CH_4 emissions (Ripple *et al.*, 2014). Greenhouse gas emissions from monogastric livestock are considerably less, although still significant, resulting mainly from CH_4 and nitrous oxide (N_2O) emissions from stored manure and fertiliser used in feed production. Increasing demand for livestock feed is also driving emissions, and deforestation for soy production in the global south is the largest single contributory factor in emissions associated with livestock production (Steinfeld *et al.*, 2006).

The contribution of livestock to global warming is set to increase as world populations and food demand continues to grow, and there is a need for more efficient production and consumption by global populations (Smith, 2013). Some commentators have called for a shift towards lower-meat diets, but it is recognised that a complete conversion to vegetarian or vegan diets could have unintended consequences (Garnett, 2009; Hallström *et al.*, 2015). Livestock systems provide a range of ecosystem services, ranging from aesthetic value to employment (Chatterton *et al.*, 2015; Rodríguez-Ortega *et al.*, 2014), and livestock may be grazed on land unsuited for crop production. Soil carbon sequestration under ruminants on such land, through improved grassland management and carbon cycling in manure deposition, may be an effective greenhouse gas mitigation option if the net gains exceed increases in CH_4 emissions, but there is a finite upper limit to sequestration in permanent grassland. Developments in this area are pertinent to the 4 per 1000 initiative to promote soil carbon sequestration (Minasny *et al.*, 2017). The role of mixed crop-livestock farming systems in improving environmental efficiencies has also been emphasised within studies exploring the relative performance of specialised and integrated crop-livestock systems, with recent trends towards the separation of these components in agricultural systems exemplifying the need for more integrated approaches (Lemaire *et al.*, 2014; Wilkins, 2008).

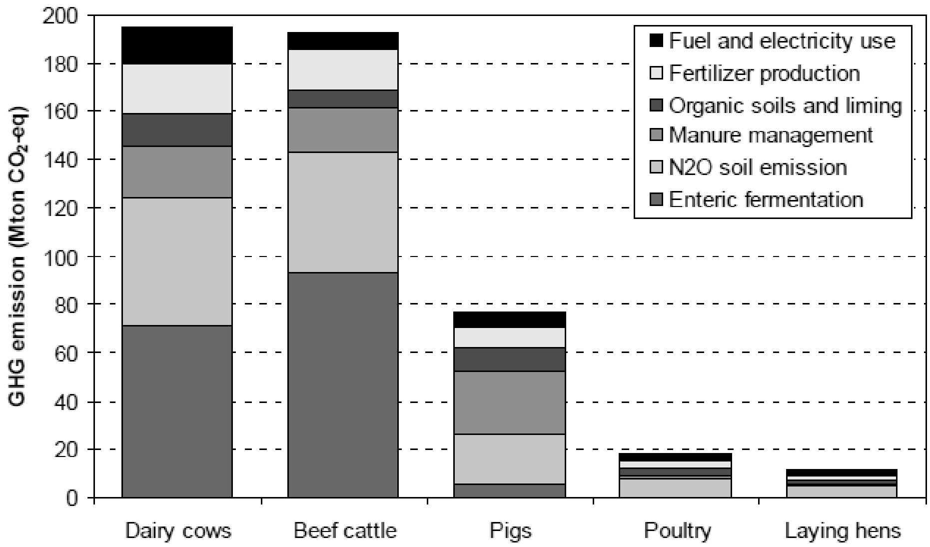


Figure A.1: Greenhouse gas emissions from the various emission sources associated with the dominant forms of livestock production in the EU-27. Emissions caused by direct or indirect land use change, such as deforestation in Brazil or conversion of pasture and scrubland in Argentina, were not included given the complexity of the processes, drivers and sectors involved. Source: Lesschen *et al.* (2011)

Within this setting, organic farms represent an approach to production that places a special emphasis on reduced inputs (e.g. with regard to concentrate feed and crop protection products) and mixed farming (Lampkin *et al.*, 2015). These approaches are enforced by organic standards which have been developed in-line with four key organic principles defined by the International Federation of Organic Agriculture Movements (i.e. Health, Ecology, Fairness, Care, IFOAM, 2005). As a result of this emphasis the diverse range of approaches applied on many organic farms can lead to improvements in resource-use efficiency, as a result of reduced inputs per unit of output, and increased soil organic carbon (SOC) concentrations in arable soils (Reganold and Wachter, 2016). Despite these benefits, the environmental performance of organic livestock production relative to conventional/non-organic is still a matter of some debate, and can depend greatly on the livestock type and the unit of comparison (Lampkin *et al.*, 2015). For most organic livestock products, reduced inputs per hectare result in lower greenhouse gas and fossil energy use per unit of land area, whereas impacts per unit of product can be worse (see Figure A.2 and Figure A.3). This is particularly so for monogastric livestock, where the requirement for essential amino acids presents a particular challenge because, unlike ruminants, pigs and poultry are unable to produce their own and certified organic producers are unable to feed the synthetic amino acids commonly used in conventional systems (van de Weerd *et al.*, 2009). Difficulties in this area contribute to less efficient feed conversion and increased mortality rates in organic poultry production and can increase environmental impacts per unit of product compared to full-housed or free-range systems (Leinonen *et al.*, 2012a, b). Similarly pig production systems can perform worse as a result of inappropriate breeds, lower stocking densities and less output per hectare (Van der Werf *et al.*, 2007). Reduced milk yields, longer rearing periods, and higher feed conversion ratios on organic dairy farms can also lead to worse performance per litre of product (de Boer, 2003; Williams *et al.*, 2006). Additionally, it has been suggested that the lower yields from organic farming may result in increased agricultural land-areas, and greater impacts on biodiversity (Green *et al.*, 2005) although more intensive, higher yielding measures suggested as an alternative may also lead to expansion of agricultural areas as a result of market-forces (e.g. lowered commodity prices that result from over-production can encourage increased areas of production to ensure a viable economc return, Perfecto and Vandermeer, 2010).

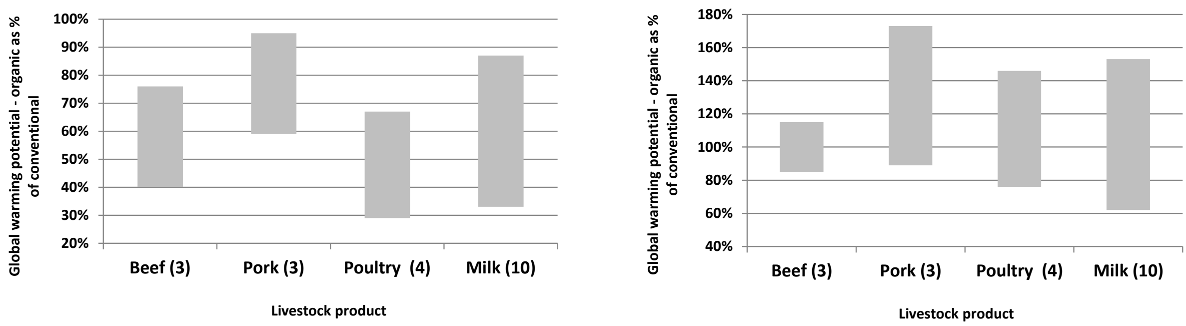


Figure A.2: Results from a review of Life Cycle Assessment (LCA) studies comparing the global warming potential of organic/non-organic production. Impacts of organic production are expressed as a percentage of non-organic per unit of area (left) and per unit of product (right). Values in parenthesis refer to number of studies within each product category. Adapted from Meier *et al.* (2015)

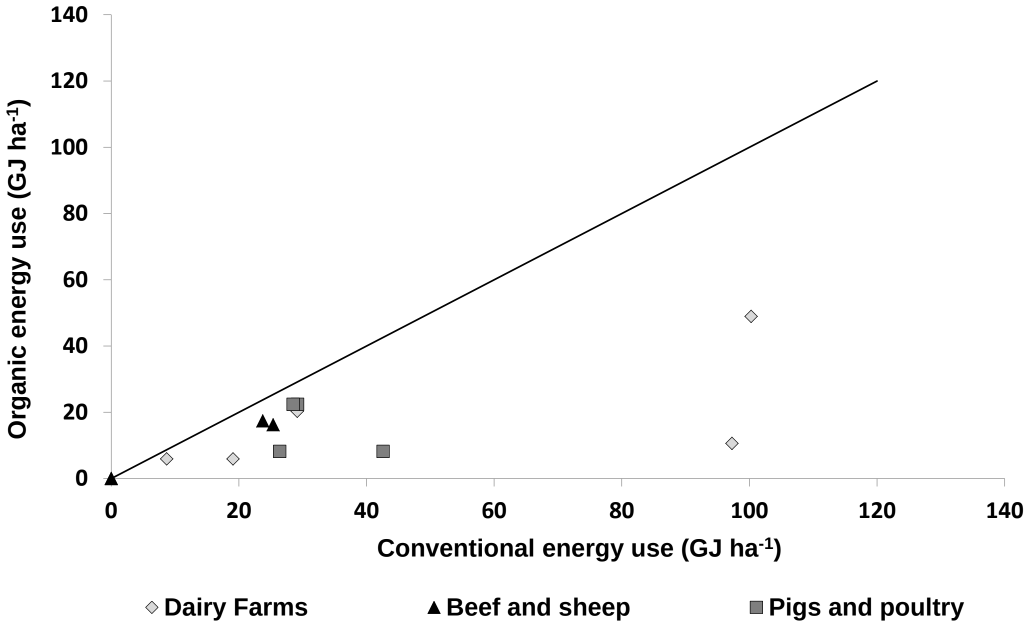
Although organic systems can therefore provide useful examples of improved environmental performance, the extent of the difference can depend greatly on the systems being compared (in particular the system boundaries), data sources and variation in environmental factors and/or typical practices between countries or regions. This chapter will provide examples of this variation and highlight particular challenges and innovative solutions identified within the sector. Examples of innovation in practice will also be given through case study portraits of producers already adopting measures that are leading to gains in environmental efficiency with respect to greenhouse gas mitigation and fossil energy use.

A.3 Greenhouse emission mitigation and energy efficiency in organic farming

This section gives examples of greenhouse gas mitigation and improved energy efficiency through organic livestock farming, and highlights particular challenges for the sector with regard to livestock feeding, manure-management and farm system design.

A.3.1 Improved feeding for greenhouse gas mitigation in organic livestock farming

As outlined above, organic systems focus on reduced inputs to create resilient and sustainable systems. This approach distinguishes organic farming from other modes of production that focus on single aspects. In practice, this systems approach leads to the application of production methods that can encourage a good biological balance and create a self-regulating farm with respect to livestock feed supply and demand, although it is recognised that this often cannot be absolutely attained (Lampkin *et al.*, 2015). In livestock farming, these approaches lead to adoption of practices that endeavour to meet physiological needs of animals whilst reducing environmental burdens created through the use of imported feed. The production of meat/milk from forage is therefore a central tenet of the organic approach, and the EU organic standards dictate that at least 60% of the diet (on a dry matter basis) for ruminants shall be forage based. Although many European cattle production systems are already adopting forage-based diets, an emphasis on pasture-fed livestock within organic production limits the use of crops for feed, and lower concentrate-feed rates are generally found in organic ruminant livestock systems (Lund and Algers, 2003).



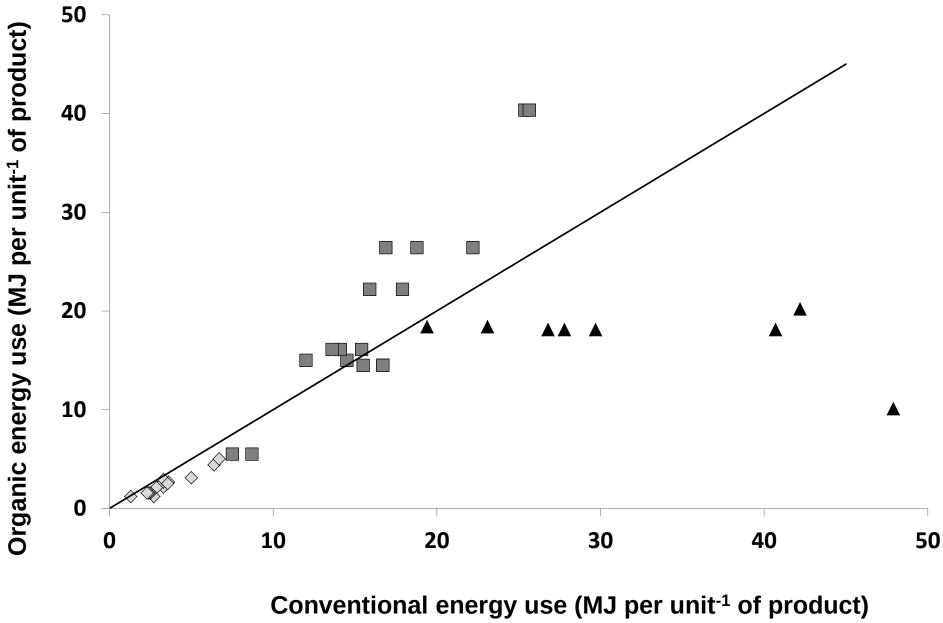


Figure A.3: Fossil energy efficiency of organic and conventional livestock production - results from 21 comparative studies. Organic performs better below the line, worse above the line. Note the 'trend-line' is x=y for the purposes of illustrating the relative performance for each product type and is not a line of best fit. Production units were not constant across the studies compared. Adapted from Smith *et al.* (2014)

This approach limits the amount of resource used in the production of feed crops, which currently poses a substantial challenge to global warming and food security

(33% of global arable land is used for livestock feed production, Ripple *et al.*, 2014) and contributes to land clearing and subsequent soil degradation (El-Hage Scialabba and Müller-Lindenlauf, 2010). The common use of clover and other nitrogen-fixing legumes in temporary grassland within organic systems also allows for the avoidance of manufactured nitrogen fertiliser, and the associated fossil energy input, leading to substantial improvements in production energy efficiency per unit of product (see Figure A.3) whilst supplying the ruminant animal's protein and energy consumption requirements in a manner that can help to promote improved animal health (Lund and Algers, 2003).

Organic production methods can therefore help to offset the impacts associated with the recent growth in feedlot-based production, which are particularly evident in South American countries such as Argentina and Brazil (Deblitz, 2012; Malau-Aduli and Holman, 2014), areas where emissions from deforestation are already particularly relevant (Deblitz, 2012; Flysjö *et al.*, 2012; Malau-Aduli and Holman, 2014) although it should be noted that high concentrate-feed systems can still outperform organic production in environmental terms, as a result of increased outputs, better weight gain efficiency and improved manure management (Peters *et al.*, 2010; Ross *et al.*, 2014). The reliance on forage in organic systems can also lead to higher CH_4 emissions per kilogram of product, as a result of increased dry matter intakes and reduced digestibility compared to systems feeding high-levels of concentrate (Williams *et al.*, 2006) although others note the severe difficulties relating to health and longevity that can occur in cereal/concentrate intensive systems (e.g. through increased incidence of acidosis) which may, paradoxically, lead to an increase in herd size and greater emissions overall (Novak and Fiorelli, 2009). Such issues are less likely to be present in non-organic grassland-dominated systems, however the increasing popularity of cereal based beef production at a global level, makes the increased resource-use efficiency of less input-intensive approaches an important consideration (Deblitz, 2012; Lynch *et al.*, 2011; Smith *et al.*, 2015). Lower replacement rates on organic dairy farms can also lead to reduced greenhouse gases if dairy calves produced are used for beef production, with the increased longevity of organic herds reducing or offsetting the emissions associated with the unproductive rearing of dairy cows and the need for suckler cows (Flysjö *et al.*, 2012; Idel, 2013) although this is dependent on there being demand for milk (the supply from suckler beef and associated GHGs would increase, if the demand for milk falls and the demand for meat increases).

As outlined above, organic pig and poultry production can also perform worse than conventional systems, in terms of greenhouse gas and fossil energy efficiency, as a result of poorer feed conversion. In particular this relates to the inability of monogastic animals to produce their own amino acids as part of the digestion process. Although non-organic production systems can overcome this through supplementation with limiting synthetic amino acids (in particular methionine and lysine) their use is currently prohibited in organic systems. An imbalanced supply of amino acids in organic poultry production can therefore result, despite the high protein concentration of soy and approved oilseed meals, as they are still deficient in essential amino acids. In addition to affecting productive performance, this approach can lead to increased nitrogen output in excreta, through overfeeding of protein, and increased nitrogen losses in the outdoor run (Steenfeldt and Hammershøj, 2015). Higher mortality rates can also occur in organic systems as a result of increased metabolic energy requirements, predation pressures and greater incidence of feather pecking as a result of untrimmed beaks (Dekker *et al.*, 2012). Similarly the use of high protein feedstuffs in organic pig production systems can result in increased nitrogen losses, whilst limited amino acid supply, increased occurrence of coccidiosis and internal/external parasites through access to an outdoor area reduces feed efficiency and increases N and N_2O losses via leaching and denitrifcation (Edwards, 2005; Halberg *et al.*, 2010; Hovi *et al.*, 2003; Strid Eriksson *et al.*, 2005).

Performance in this area could be improved by making better use of the range to supply nutritional requirements, thereby reducing the need for imported feed and excessive protein inputs. Recent work completed within the Core Organic 2 project Improved Contribution of local feed to support 100% Organic feed supply to Pigs and Poultry (ICOPP) highlighted that herbage can meet 50% of the maintenance energy of dry sows, with lucerne and other legumes representing particularly promising crops due to their high protein, lysine and methionine contents (Crawley, 2015). Alternative crops such as quinoa also offer potential for improving feed efficiencies in organic farming with particular regard to the balanced supply of limiting amino acids (Steenfeldt and Hammershøj, 2015). A study of six poultry meat farms in central Italy also highlighted the potential benefits of slower growing strains, which can better utilise the natural environment through foraging behaviour. The same study highlighted the need for a greater emphasis on the benefits provided by ranging areas with regard to ration formulation (Castellini *et al.*, 2012). A lack of suitable breeds hinders developments in this direction however, and improvements in this area are likely to be necessary for environmental and animal welfare improvements to be realised within the organic sector (van de Weerd *et al.*, 2009).

Inbalanced diets can also present an issue for ruminant livestock, in particular when animals consuming lush pastures take in amounts of protein far exceeding their requirements, which in turn leads to increased N_2O loss following excretion (Eckard *et al.*, 2010). Balancing the high protein forages often found in organic systems (e.g. red clover and lucerne) with other feedstuffs that have a higher energy-to-protein ratio (e.g. maize or cereal silages) or providing high-energy supplements (e.g. concentrates and sugar processing by-products) could help to reduce losses associated with high protein diets in organic feeding regimes, although a balance needs to be achieved in order to minimise consumption of human-edible components (Dijkstra *et al.*, 2011; Eckard *et al.*, 2010). A lack of availabity of organic low protein/high energy processing residues (e.g. sugar beet pulp, brewers grains) as a result of the small size of the sector also limits developments in this area. Breeding animals with improved nitrogen use effiency could also help to foster improvements in nitrogen efficiency, and the use of older animals in dairy farming may help to reduce N excretion per kg of milk, as a greater proportion of the protein consumed is used for milk production, as opposed to maintenance (Børsting *et al.*, 2003).

A.3.2 Manure and slurry management

The development of healthy, stable and fertile soils is a key objective of the organic approach. Use of farmyard manure is therefore a key element of organic farming systems (Lampkin *et al.*, 2015). Whilst this approach avoids the emissions and fossil energy use associated with fertiliser manufacture, and can lead to increased soil organic matter (SOM) contents, it can result in greater impacts on the environment compared to the use of mineral N fertiliser as a result of difficulties in synchronising N availability with crop demand, resulting in nitrate-N leaching, and, in wet soils, N_2O emissions due to high concentrations of N and organic C together (Rodrigues *et al.*, 2006). Furthermore, the application of manure incurs high ammonia emissions and is more difficult to use with low loss application systems compared to slurry. The deep litter animal bedding approach commonly applied on organic units can also contribute to greater losses from the system through N_2O release as a result of anaerobic conditions created through compaction by animals (Chadwick *et al.*, 2011). Deep litter bedding systems may also result in greater amounts of CH_4 compared to slurry systems, due to increased temperatures in the deep litter stack and compaction from animals which results in anoxic conditions and degradation of organic matter (Monteny *et al.*, 2001). Moving towards slurry based systems within the sector could therefore help to improve performance with respect to greenhouse gas mitigation - particularly where slurry stores are covered - although such a shift may present a conflict with the organic principles and standards which prescribe minimum requirements for the provision of dry-litter and non-slatted floor areas in housing to reduce stress (Chadwick *et al.*, 2011; Novak and Fiorelli, 2009). Applying anaerobic digestion on organic farms could also help to improve nitrogen use efficiency, by providing a readily available-N source that can help to meet crop demands at times of peak demand whilst reducing the CH_4 emissions associated with the storage of manure (Novak and Fiorelli, 2009). The benefits in terms of N efficiency are maximised with the use of low-loss application methods. Anaerobic digestion, however, converts organic matter into biogas so that less is available to the soil, and although the biogas also offsets the need for fossil energy, the use of such approaches could be seen to be in conflict with core organic principles, which give emphasis to the importance of developing long-term soil health instead of feeding plants directly. Such practices may also encourage the presence of nitrophilious weeds on the farm through increased nitrogen availability (Stinner *et al.*, 2008).

A.3.3 Integrated approaches and sustainable landscape design

As mentioned above, the greater diversity in organic farming systems can lead to a more integrated production system that allows farmers to spread manure produced by livestock on cropping areas within the same farm. This can help to avoid the stockpiling of nutrients in the form of manure and slurry, often seen in conventional livestock farms, which can lead to greenhouse gas emissions and other environmental problems (El-Hage Scialabba and Müller-Lindenlauf, 2010). In addition a more holistic and integrated approach can help to reduce the climate-related impacts resulting from the production of animal feed, by ensuring that a greater proportion of animals' nutritional requirements are met by forage produced on-farm (Idel, 2013). At the same time the use of ruminants in farming systems can help to improve degraded land through manure returns, and improved rotational grazing practices. The associated gains in SOC represent an important greenhouse gas mitigation measure that could be applied across a range of agri-climatic zones to create lower-impact, resilient and adaptive systems (Idel, 2013; Smith, 2014). It is also important to consider the benefits that such integrated systems provide in terms of converting forage to edible food, and the current positioning of ruminants as poor convertors of feed seems to overlook the human-edible component of livestock diets in determining feed conversion efficiencies (Wilkinson, 2011). Wilkins (2008) also highlights the benefits that could be obtained through a widespread adoption of integrated approaches, in terms of nutrient and energy-use efficiency, and in particular the potential role of 'mixed farming at a distance' (i.e. transfer of manures from specialised livestock to cropping farms). Although many organic farms are already applying such integrated approaches, the transfer of manure between cropping and livestock farms could become increasingly relevant to the sector, if current trends towards specialisation continue (Guthman, 2014).

Whether the mixed approaches used in organic production always offers a more efficient alternative when such aspects are considered is still a matter of debate, particularly given the environmental potential of integrated approaches that adopt both organic and non-organic practices (Reganold and Wachter, 2016). In addition there is certainly scope for improving performance in complex mixed crop and livestock (e.g. through model-based support and multiple-objective optimisation, Groot *et al.*, 2012) and organic systems may fail to reach to economy of scale realised through more intensive, specialised production systems. However the flexibility inherent in mixed crop-livestock systems typically found within the sector constitutes a real strength that could allow for greater long-term environmentally sustainability in livestock production, in the face of fluctuating climatic conditions (Altieri *et al.*, 2015).

The diverse, systems approach applied in the organic sector can also offer improved performance when considering a range of ecosystem services, and it has been suggested that the reduced outputs from organic and agroecological systems may represent a 'sustainable optimum' when broader environmental, economic and social objectives are considered (e.g. with respect to resilience to extreme weather events and human health, Altieri *et al.*, 2015; Reganold and Wachter, 2016). With these broader elements in mind there have been calls to move away from assessments that focus on single agricultural products and individual environmental impacts categories such as greenhouse gas emissions, towards system-level comparisons that utilise a range of qualitative and quantitative assessment methods to capture externalities and trade-offs between multiple dimensions such as soil, human health and the effects of adopting new technologies (Garnett, 2014). There have also been calls to consider the role of waste reduction and human diets in improving the sustainability of food systems, with a recent study suggesting that the widespread implementation of demand-side measures could reduce food-related greenhouse gas emissions by approximately 45%, compared to a 2050 baseline (Bajzelj *et al.*, 2014). Another recent study has highlighted that diets that have a low greenhouse gas impact are likely to be in-line with dietary guidelines for most nutrients, suggesting a possible synergy between human health and greenhouse gas mitigation (Bälter *et al.*, 2017). Affecting behavioural change towards healthy and sustainable diets presents a challenge however, and an integrated approach of economic incentives and improved information provision is likely to be required for real progress to be made in this area (Bajzelj *et al.*, 2014; Macdiarmid *et al.*, 2011).

A.3.4 Organic livestock systems and carbon sequestration

As outlined above, livestock production is the largest contributor to agricultural land-use, and if all land used for livestock production were used for carbon sequestration, 25–470% of the greenhouse gas emissions associated with food production could be offset (Ripple *et al.*, 2014; Schmidinger and Stehfest, 2012; Smith *et al.*, 2013). At the same time livestock can indirectly contribute to increased soil carbon concentrations, in particular through manure deposition and improved grassland management (Smith, 2004) although management options that increase soil carbon concentrations may reduce outputs per hectare, and subsequently lead to increased demand for land and emissions from land-use change following agricultural expansion (Paustian *et al.*, 2004; Powlson *et al.*, 2011; Smith *et al.*, 2013). Carbon sequestration in agricultural soils is also finite, as soils will reach a new steady state over time, and the process is reversible (i.e. the majority of any carbon that is built up over time will be released following cultivation, Powlson *et al.*, 2011). Full-greenhouse gas accounting should also be applied when considering the impacts of individual practices, as some measures that increase soil carbon accumulation may result in increased emissions from other sources (e.g. increased N_2O emissions from fertiliser use, Paustian *et al.*, 2004).

Despite lower yields in organic livestock production, a meta-analysis conducted in 2010 confirmed a 2.2% average annual increase in soil carbon contents within organic systems as a result of increased amounts of organic fertiliser (i.e. livestock manure and/or composts) and diverse rotations incorporating fertility building leys (Leifeld and Fuhrer, 2010). A review of field studies carried out in the US also found that legume and manure based organic management resulted in similar levels of SOM increase and that the ley period alone is more significant than additions of manure, in terms of soil carbon contents (Marriott and Wander, 2006). Improved rotational grazing practices also have the potential to sequester carbon by encouraging productivity and below-ground biomass, particularly on degraded land, although the benefits that can be obtained in this area may have been exaggerated in some cases (Nordborg, 2016).

A study carried out by the Swiss Research Institute of Organic Agriculture (FiBL) built on the work by Leifeld and Fuhrer (2010) by reviewing 74 studies to identify differences in SOC sequestration under organic and non-organic management. The results from this work revealed significantly higher SOC concentrations (+0.18±0.06% points), stocks (3.50 ±1.08 Mg C ha^{-1}) and sequestration rates (0.45 ± 0.21 Mg C ha^{-1} yr^{-1}) compared with non-organic management (mean values ± 95% confidence interval). As with the study by Leifeld and Fuhrer (2010), the prime cause was found to be the use of ley/arable rotations and the application of organic fertilisers (Gattinger *et al.*, 2012). Despite this observed benefit, Leifeld and Fuhrer (2010) highlight that the amount of organic fertiliser used in many organic systems generally exceeds that applied in conventional systems, and as the manure will generally be applied to agricultural land, organic systems cannot be considered to provide a net benefit. Leifeld and Fuhrer (2010) also state that a truly unbiased comparison should be based on similar organic fertiliser application rates, and crop rotations incorporating fertility building leys, as neither of these aspects are unique to organic farming systems. Whilst this is true, an experiment of this kind would lose the significance of the farming system. In reality organic farmers are more likely to be using manures and a fertility building periods in their crop rotation as European organic regulations dictate that the fertility of the soil should be maintained and increased through crop rotations including legumes, and through application of such materials. 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 in order to create balanced systems. In addition, when Gattinger *et al.* (2012) limited the assessment to organic farms receiving zero net-inputs, significant, positive differences in SOC concentrations and stocks were still found (0.13 ± 0.09% points and 2.16 ± 1.65 Mg C ha^{-1}, respectively) although differences for sequestration rates were no-longer statistically significant (0.27 ± 0.37 Mg C ha^{-1} yr^{-1}).

As mentioned above any increase in soil carbon concentrations resulting from the adoption of such practices is finite and is likely to occur within the first 20 years following a shift in land-management, as a new steady-state is reached (Gattinger *et al.* 2012 found that differences in sequestration rates between the systems were only significant within the first 20 years of conversion). Long-term studies have also highlighted that the rate of SOC accumulation over time is non-linear and will differ according to the previous land use and the amount of organic material that has been applied. For instance the 140-year Broadbalk Experiment at Rothamsted Research, UK, found that on the 'farmyard manure' plots the rate of increase was greatest in the early years of application, and reduced over-time as the soil approached a new state of equilibrium (Powlson *et al.*, 2011). It is therefore critical that the initial, large gains in soil carbon sequestration, resulting from the addition of manures or other organic materials, are not extrapolated year on year under an assumption that the same increase will be observed indefinitely (Powlson *et al.*, 2011). This view is supported by Smith *et al.* (2007) who highlight that terrestrial biomass sources only remove carbon from the atmosphere until the maximum capacity for the ecosystem is reached – which may take 15 to 33 years, according to the management practice and farming system. This phenomenon is referred to as ‘sink saturation’ (see Figure A.4).

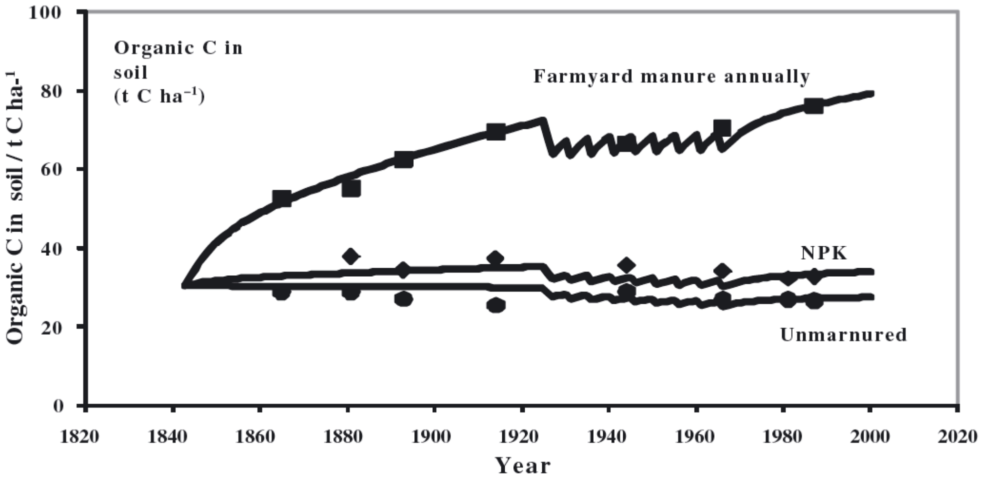


Figure A.4: The accumulation of total organic carbon in the topsoil (0-23cm) of silty clay loam soils at Rothamsted, UK, under three treatments: no fertilizers or manure applied since 1844; P, K, Mg plus 144 kg N ha^{-1} since 1852; farmyard manure at 35 t ha^{-1} fresh weight applied since 1885 plus 96 kg N ha^{-1} since 1968. Source: Powlson *et al.* (2011). Amount of manure applied (35 tonnes per hectare) is equivalent to manure generated by 5 x beef suckler cattle over 6 months housed period – author’s calculation based on Defra RB209 (Defra, 2010)

The concept of a saturation point is challenged somewhat by Soussana *et al.* (2007) who carried out an assessment of the greenhouse gas budget of nine European grassland sites using an eddy-covariance method which measures the Net Ecosystem Exchange (NEE) of carbon at a fine temporal resolution (see Gilmanov *et al.*, 2007 for more details on the approach). Losses of carbon as CH_4 were also recorded *in situ* across all sites, whilst imports and exports of organic-C as manure, animal body-mass and harvested plant biomass were recorded and accounted for on-top of the NEE. The study therefore provides a comprehensive assessment of C imports and exports of a range of permanent and temporary grassland sites (Smith, 2014). The results revealed that established grassland sites showed a large sink potential through a negative carbon balance (i.e. carbon imports to the grassland system exceeded exports) whereas the newly sown grass-clover mixtures displayed a net loss for one year out of two. This challenges the conventional wisdom that permanent grassland systems tend to be at an equilibrium value (i.e. at the sink-saturation point referred to in Figure A.4) and that newly-sown grasslands can store more carbon than established pastures (Soussana *et al.*, 2007) although others have noted that perpetual increases in soil carbon concentrations under long-term/permanent grassland are untenable given results from long-term trials/surveys (Smith, 2014).

A.3.5 Agroforestry in organic livestock and GHG mitigation

Farming systems incorporating trees and livestock can also encourage improved greenhouse gas mitigation, and the uptake of these practices is encouraged within different standards for organic agriculture. In particular, the adoption of these systems has the potential to reduce pressure on existing forests and can improve carbon sequestration in agriculture through the generation of above and below ground biomass (Lampkin *et al.*, 2015). Improved efficiency of livestock production can also be encouraged through the uptake of agroforestry, in particular through improved weight gains from increased shade, which limits high body temperatures and the associated appetite suppression (Smith *et al.*, 2012) although the trees occupy land that could be used for forage/feed production. Improved nitrogen use efficiency in feed crop production can also be encouraged through deposition of leaf litter by tree species that produce high amounts of leaf biomass (e.g. hybrid poplar), which can reduce the need for mineral fertiliser and the associated N_2O emissions (Thevathasan and Gordon, 2004) and decreased nitrogen leaching in agroforestry systems may have the potential to reduce N_2O from denitrification in surface water (Smith *et al.*, 2012). The ammonia (NH_3) abatement potential of agroforestry systems has also been highlighted through a recent project which illustrated the benefits of enhanced tree cover in agricultural landscapes (Bealey *et al.*, 2013). Despite the benefits that accrue in these areas, current uptake of agroforestry in livestock systems is low and a lack of targeted financial support is hindering the development of the sector in many countries. In the present climate uptake may therefore be limited to less intensive farms and those within the organic sector.

A.4 Examples of Innovation in Practice: livestock farmers progressing towards greenhouse gas mitigation

In this section, examples of innovative farmers operating in the UK are given to provide an overview of how greenhouse gas mitigation measures are being applied to improve performance. Each of the farm businesses described are already applying research in practice to achieve progress towards environmental and economic efficiency and each represents an example of farmer-led system development.

A.4.1 Case study 1: Organic Dairy Farmer in Shropshire: incorporating trees with livestock

Tim Downes runs a 300 cow spring calving dairy herd in the west of England and is currently making good use of trees on his land to obtain benefits in the following areas:

- N leaching - trees are helping to keep vital nutrients in the field, potentially helping to reduce indirect N_2O emissions from denitrification
- Improved grass growth in the spring - trees are providing increased shelter and temperature regulation, potentially helping to improve pasture productivity
- Renewable energy - the farmhouse is now completely self-sufficient for woodfuel harvested from the on-farm woodland and shelterbelts

A recent project has involved Tim planting a dedicated area of sycamore, hornbeam, lime and elm as a nutritional trial which will eventually supplement the herd's nutrition. A separate paddock consisting of crack and white willows will also be used to investigate the potential benefits that may be provided through anti-inflammatory properties provided by salicylic acid in the willow trees, which could help cows with mastitis and other conditions. Tim will also be investigating whether the trees help to improve parasite control on the farm by increasing the tannin content of the livestock diets.

Figure A.5: a new agroforestry project has been established at Tim Downes's organic dairy farming operation in Shropshire, UK, which will investigate the potential benefits that trees may provide in terms of nutrition and livestock health

A.4.2 Case study 2: The Lakes Free Range Egg Company: encouraging better use of the range through silvopoultry

Tree planting has also been used to promote environmental benefits at the Lakes Free Range Eggs Company in Cumbria where over 200,000 trees have been planted over 2400 hectares. Benefits observed include:

- Improved egg quality (i.e. increased percentage of class A eggs) encouraged through reduced stress and increased ranging behaviour (Bright and Joret, 2012)
- Reduced mortality as a result of increased cover - trees protect against predation and climatic extremes through canopy cover whilst encouraging ranging outside which reduces indoor stocking rates, feather pecking and mortality rates (Bright *et al.*, 2011; Bright and Joret, 2012)
- Carbon storage in above and belowground biomass (Bright *et al.*, 2011).

Fast growing species such as poplar and willow have been combined with slower growing natives such as oak and maple to create a highly biodiverse approach to range management. Whilst trees in the range cannot replace the need for bought in feed, David Brass, CEO, believes that they play a valuable role in promoting the welfare of animals and that the associated business case is strong, with the results in terms of improved product quality more-than paying off the cost of establishment.

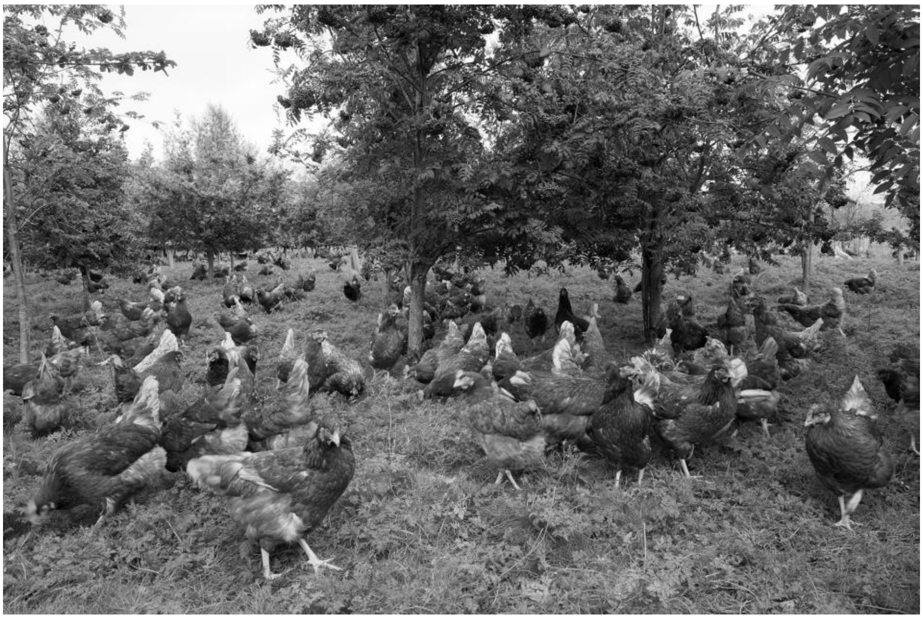


Figure A.6: a diverse mix of trees at The Lakes Free Range Egg Company. The trees encourage better use of the range, improve product quality and reduce mortality rates whilst promoting carbon sequestration and helping to offset the emissions associated with feed imports.

A.4.3 Case study 3: Fordhall Farm: 100% pasture fed livestock

As outlined above moving towards grass based diets is a central tenet of the organic approach, and this method is exemplified by the 100% forage system being applied at Fordhall Farm in Shropshire. Arthur Hollins developed the “foggage” feeding approach where a diverse grassland mixture accumulates enough grass in the growing season to sustain cattle outdoors all year round. Benefits of this approach include:

- Very low fossil energy use in forage production as animals graze all-year (as some of the soils cannot support all-winter stocking, livestock are moved to sandy hills during wetter months and to the wetlands during the dry summer months)
- Reduced risk of poaching through the tight root structure of the highly-diverse permanent grassland

- The permanent grassland also provides a substantial carbon sink - although the soil carbon accumulation rate is likely to be low as the grass was established over 65 years ago

Although the 100% forage system reduces the costs and environmental burdens associated with feed inputs, slower growing cattle will emit more CH_4 per unit of liveweight gain.

Figure A.7: diverse, herbal leys established over 70+ years are an essential part of maintaining a 100% forage system at Fordhall Farm

A.4.4 Case Study 4: Pound Farm: building soil organic matter through composting and effective rotation design

The maintenance and improvement of soil health is central to the organic approach and practices that can promote developments in this area have been applied on Pound Farm, in South West England for over 15 years. Grass clover leys are used on over 30% of the cultivated area, and composted manure and imported green-waste compost have been regularly applied since the farm converted to organic status. Whole crop cereal silage has also been used to provide for the cattle's nutritional requirements and to reduce imported feed usage. Potential benefits of these practices include:

- Reduced energy and greenhouse gas burdens from the production and transport of imported feed

- Reduced N excretion through a balance of protein and energy in livestock rations
- Improved soil carbon sequestration over time (see Figure A.8)
- Improved soil biological life

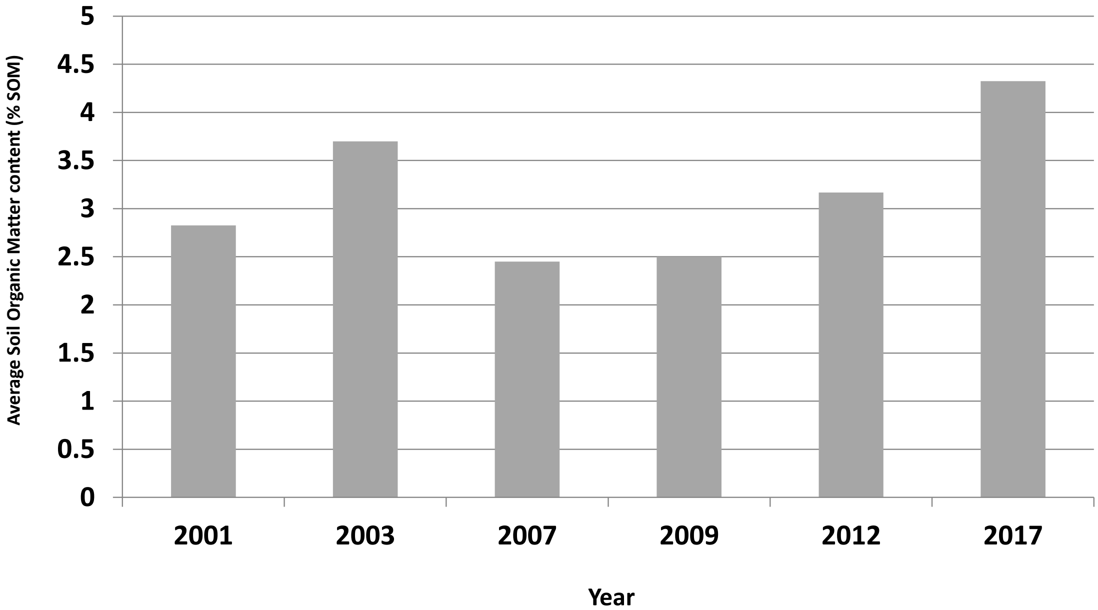


Figure A.8: Soil Organic Matter (SOM) records from Pound Farm collected over 16 years (average data from 4 fields)

A.5 Challenges and opportunities in research and development

In spite of the mitigation and carbon sequestration benefits that can accrue from the practices described in the above case-studies, accounting for the greenhouse gas saving that can result from their adoption presents a challenge. In particular there is a divergence over the unit of comparison that should be applied when comparing individual practices and system-level approaches such as organic management. Although many commentators suggest that expressing impacts per unit of food produced is the most logical approach, in view of the need to produce more from less resource (Leifeld and Fuhrer, 2010) others highlight that focusing on impacts per unit of product can create a “blind spot” by hiding the increased impacts associated with imports and the additional land-use associated with their production (Salou *et al.*, 2017) although this does not occur when authentic life cycle assessment is applied in the analysis. It has therefore been suggested that the focus should shift from simple production factors and instead consider how land management can be adapted to promote healthy and sustainable economies (McIntyre *et al.*, 2008). Whilst combining greenhouse gas and fossil energy assessments with ecosystem service evaluation can provide a useful method of capturing these broader elements (Chatterton *et al.*, 2015) further developments are required in this area to effectively quantify performance and ensure that trade-offs are minimised.

The importance of the long-term impacts of contrasting farming systems has also been highlighted in the context of climate change research and the predicted increases in drought and flooding (Altieri *et al.*, 2015). In this respect there is evidence that organic farming can provide a more resilient system, in particular through improved soil health and associated water retention (Lampkin *et al.*, 2015). Challenges are also faced in research on the costs/benefits associated with changes in soil carbon stocks within farming systems. Although it is possible to account for soil carbon sequestration gains on a unit of area or country basis, the non-linear nature of any increase, site specific variation, and various timeframes used as a basis for the as assessment, can make it difficult to express benefits per unit of agricultural product on a consistent basis (Petersen *et al.*, 2013).

Research also has a key role to play in the development of more sustainable farming practices within the organic livestock sector. In particular, adapting monogastric breeds for organic management (i.e. developing breeds suited to longer rearing periods and increased reliance on feed foraged from the range and/or using dual purpose breeds) would help to reduce the high metabolic pressure created by combining organic diets with breeds more suited to intensive conventional systems, and other factors conflicting with the IFOAM organic principles (e.g. discard of male chicks in egg production systems, van de Weerd *et al.*, 2009). It is also clear that organic management practices could play a key role in reducing greenhouse gas emissions within the non-organic agriculture sector, and integrated approaches to livestock management can offer a highly efficient approach to production that can achieve some of the benefits of lower-inputs without sacrificing as much productivity. From a regulatory perspective, adjusting the organic standards could also allow for improvements in the relative performance of the monogastric livestock sector, in particular as higher mortality rates and disease/predation pressures are associated with outdoor systems (van de Weerd *et al.*, 2009; Weeks *et al.*, 2016). As part of the

IFOAM-led ‘Organic 3.0’ initiative, there is scope to review the role of organic standards in encouraging best practice against multiple sustainability objectives, as part of the next phase in the sector’s development (Arbenz *et al.*, 2016).

Table A.1 below summarises some of the methodological, farm-level and policy challenges and opportunities facing the organic sector that could be addressed to improve performance from a greenhouse gas mitigation and fossil energy efficiency perspective:

Table A.1. Challenges and opportunities within the organic livestock sector from a greenhouse gas mitigation and fossil energy efficiency perspective

Focus area	**Challenges**	**Opportunities**
Research	History of reductionist approaches to GHG / fossil energy use assessment (i.e. focusing on impacts for a single product) Lack of awareness on the extent to which organic / agroecological approaches can be successfully interwoven in current food systems Lack of consensus on accounting for soil carbon sequestration benefit that can be obtained from grassland / manure	Developments in farm and food system-level assessment approaches Growing interest in farmer-focussed research (e.g. through the EU Horizon 2020 programme) will help to embed greenhouse gas research in ‘real-world’ contexts Increasing interest in this area following COP 21 and the 4 per 1000 initiative
On-farm	Lower rates of productivity leading to worse performance per unit of product, particularly within poultry sector	Opportunities to improve uptake of targeted feeding measures on organic farms to increase feed conversion efficiency and reduce imports (e.g. phased feeding, sex segregation). Continuing growth of organic and free range sector may encourage further development of slow growing strains in monogastric livestock

On-farm (continued from previous page)	Manure management on many organic farms leading to high rates of N loss and 'leaky' systems Specialisation and 'conventionalization' within some organic farming systems Lack of organic feed availability with particular regard to high-quality protein and 100% organic feed supply regulation (due to be implemented in 2018)	Wider application of nutrient management planning tools to improve on-farm nutrient management and improved storage / composting Opportunities for 'mixed farming at a distance' e.g. through manure banks and use of digestate from anaerobic digestion as an effective fertiliser Developments in alternative protein and amino acid sources (e.g. insect larvae, algae)
Policy and regulative	Constraints induced by organic standards with regard to feed and nutrition (e.g. prohibition on use of pure amino acids) Lack of policy support for uptake of organic farming in some countries	IFOAM Organic 3.0 initiative recognises the importance of moving from process to outcome based indicators in ensuring the development of the sector and that this may require flexibility in standard setting Development of action plans for agroecology and organic farming (e.g. in France and Scotland) may encourage other countries to follow suit to support development of the sector and new entrants

Although many of the opportunities described in Table A.1 may be desirable from an environmental standpoint, they may also be difficult to achieve in the short-to-medium term. In particular implementation of a more flexible approach to organic standards could be a slow process in view of conflicting priorities, particularly with regard to 'no-go' areas. Encouraging greater integration of crop and livestock production systems can also present economic and logistical challenges where specialised farms are situated at some distance apart. Ensuring that stakeholders are actively engaged in research and policy developments can help to ensure efficacy of any changes whilst making sure that multiple voices are heard and end-user needs met.

A.6 Summary and outlook

Organic livestock systems can offer an improved approach to livestock production from a greenhouse gas and resource efficiency perspective, when performance is compared on a land-area basis. Differences between organic and non-organic systems are less apparent when comparisons are made per unit of product, with lower rates of production on organic farms offsetting the use of fossil-fuel intensive inputs in conventional production, particularly within the monogastric livestock sector. Although the scope to scale-up a less productive form of agriculture has therefore been questioned by some commentators, who have highlighted increasing food demands from growing populations, others highlight a critical need to adjust what we eat and the amount we waste, pointing out that changes in these areas could have a much greater impact on food security and greenhouse gas mitigation than improving the efficiency of agricultural production, and potentially allow for the broader adoption of more sustainable practices (Smith, 2013; Tittonell, 2014). The degree of complementarity between sustainable production and consumption habits is still unclear however and there is a pressing need to account for real-world complexity in farming systems, particularly in smallholder farming, on which most of the world's hungry currently rely (Horlings and Marsden, 2011). In addition, an assessment of the carbon sequestration benefits that can accrue from organic management has been entirely absent from many comparative studies, and whilst this may be a difficult area to assess, the potential benefits are such that it should not be ignored.

It is also clear that there is considerable scope for improvement within organic livestock farming. In particular ensuring that animals' diets are optimised through balanced energy and protein levels will help to avoid excessive N_2O emissions whilst potentially ameliorating the CH_4 generation associated with higher forage diets. Developments in organic breeding and range management could also help to improve resource efficiency within monogastric livestock management and greater flexibility in standard setting has the potential to address some of the challenges faced in this area. Improved manure management on organic farms could also help to improve nitrogen use efficiency whilst reducing losses of CH_4 and providing a valuable source of income. It should also be remembered that no-single farming system will be able to feed the world whilst reducing greenhouse gas emissions to a safe level, and that integrated approaches adopting the best-of-organic and non-organic approaches may offer optimal solutions. Whether such systems could be considered to be 'organic' if they adhere to the IFOAM principles is a matter of considerable debate however a flexible and innovative approach is likely to be required within both the organic and non-organic sectors if real improvements are to be scaled-up.

A.7 Sources of further information

More details on many of the issues presented here can be found on the University of Oxford and Oxford Martin Programme's Food Climate Research Network: http://www.fcrn.org.uk and through the Round Table on Organic Agriculture and Climate Change, an initiative founded in 2009 at the United Nations Climate Change Conference: http://www.organicandclimate.org

The online resource hub Agricology also provide a useful repository of information on mitigation options, carbon footprinting tools and farmer case studies: https://www.agricology.co.uk/.

Scotland's Rural College also provide a range of information sources and case studies on this topic through their Farming for a Better Climate Programme: https://www.sruc.ac.uk/info/120175/farming_for_a_better_climate

The Organic Research Centre have also devoted a section of their website to information on agroforestry, which provides details of relevant publications and links to relevant websites in this area: http://www.organicresearchcentre.com/?go=Research%20and%20development&page=Agroforestry

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Appendix B Crop groups used within Optimal Land Use Model (OLUM) and associated yields

B.1 Crop groups

Within the Optimal Land Use Model (OLUM) crops were grouped by their growth characteristics, similar nutrient requirements and pest/disease susceptibility (e.g. root vegetables, cereal crops, top-fruit) using the subset feature in GAMS[21]. Crop group categories are summarised in Table B.1 to Table B.3 below:

Table B.1: Crop groups and associated crop types – arable and pasture

Cereals	Peas and beans	Forage 1	Forage 2	Heavy feeders	Pasture
Winter wheat	Field beans	Grass/clover	Forage maize	Potatoes	Permanent pasture
Winter barley	Peas – field	Red clover	Fodder beet	Sugar beet	Rough grazing
Spring wheat	Peas – vining		Whole crop cereal	Oilseed rape	
Spring Barley					
Oats					
Triticale					
Rye					

[21] General Algebraic Modelling System (GAMS). GAMS Development Corporation. http://www.gams.com/

Within each group crops were listed as ‘stocked’ (i.e. with a manure-N requirement) or ‘stockless’ (i.e. no manure-N requirement) and by their end use (i.e. livestock feed or human consumption).

Table B.2: Crop groups – vegetables

Brassicas	Root veg	Salad	Veg legumes	Protected crops	Cucurbits	Other
Cabbage	Beetroot	Lettuce	Peas	Tomatoes	Pumpkins	Other - horticulture
Broccoli	Carrots	Spinach	Climbing beans	Cucumbers	Courgettes	
	Onions			Peppers		
	Leeks			Lettuce - indoor		
	Parsnips					
	Turnips					

Table B.3: Crop groups – fruits

Top fruit	Soft fruit
Apples	Strawberries
Pears	Other
Cider apples	
Cooker apples	

B.2 Crop yield data

Crop yields were derived through application of the NDICEA model (Nitrogen Dynamics In Crop Rotations in Ecological Agriculture) and typical organic rotations (see Table 2.2: Typical organic rotations assessed within NDICEA, in Chapter 2: Overview of Methods Applied in Thesis). NDICEA has been shown to be an accurate tool for predicting N availability in organic cropping systems on a range of soil types and rainfall zones in the UK (Smith *et al.*, 2016).

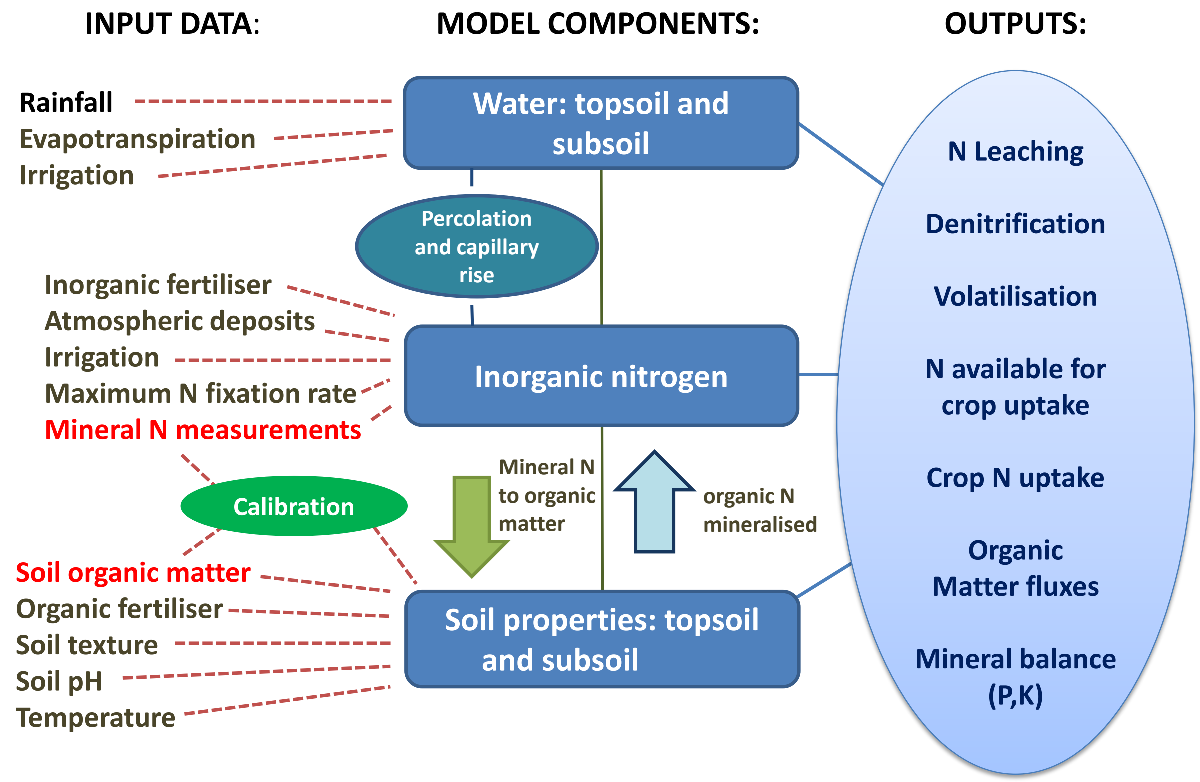

Figure B.1: Overview of the NDICEA model components and interactions

The NDICEA model consists of three modules, illustrated in Figure B.1. Within the first module, soil water dynamics are simulated taking into account irrigation, rainfall, evapotranspiration, capillary rise and percolation. The second component deals with the mineralisation of nitrogen within SOM, organic manure and crop residue. The third component deals with inorganic nitrogen dynamics, including inputs from mineralisation, atmospheric deposition, fertilisers, irrigation, biological fixation, and losses from the system in terms or denitrification, leaching and crop uptake. A calibration feature allows for automatic adjustment of 46 parameters which affect N-loss pathways such as leaching and denitrification (Swain *et al.*, 2015). The automatic calibration feature is implemented by entering soil mineral N and soil organic matter (SOM) measurements taken from the site to be assessed (Smith *et al.*, 2016; Swain *et al.*, 2015; Van der Burgt *et al.*, 2006). NDICEA aims to provide a learning tool to determine the efficacy of alternative rotation scenarios from a nitrogen use efficiency perspective and whether target yields are infeasible because of insufficient nitrogen (Van der Burgt *et al.*, 2006).

NDICEA is particularly well-suited to assessing N availability within organic systems as a result of the wide range of ley-management, cover-crop and organic fertiliser options available within the model interface. This allows for greater flexibility in designing diverse rotations and assessing their nitrogen use efficiency, compared to other tools such as PLANET (Smith *et al.*, 2009).

As the model is target-oriented, average yield input data were adjusted manually in-line with nitrogen availability over the course of a rotation, as described in Chapter 2.

B.3 Results from yield adjustments with NDICEA

Average yield data for a range of example crops are shown in Figure B.2 to Figure B.4 below:

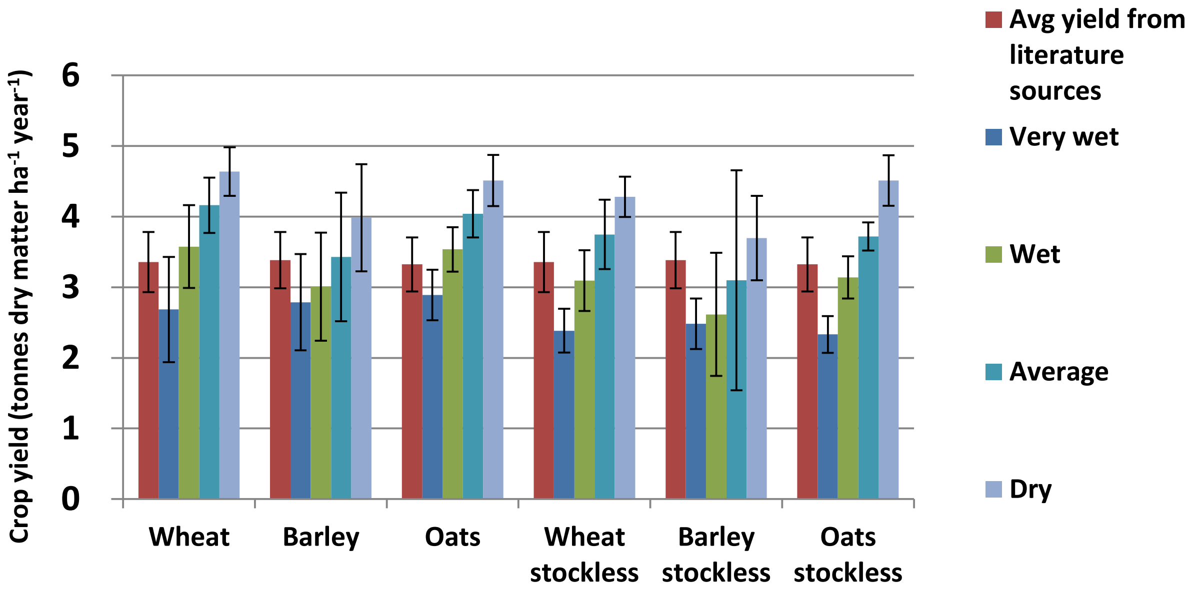

Figure B.2: Adjusted yield for organic crops grown under a range of rainfall conditions (avg. N fix scenario only) error bars = s.d.

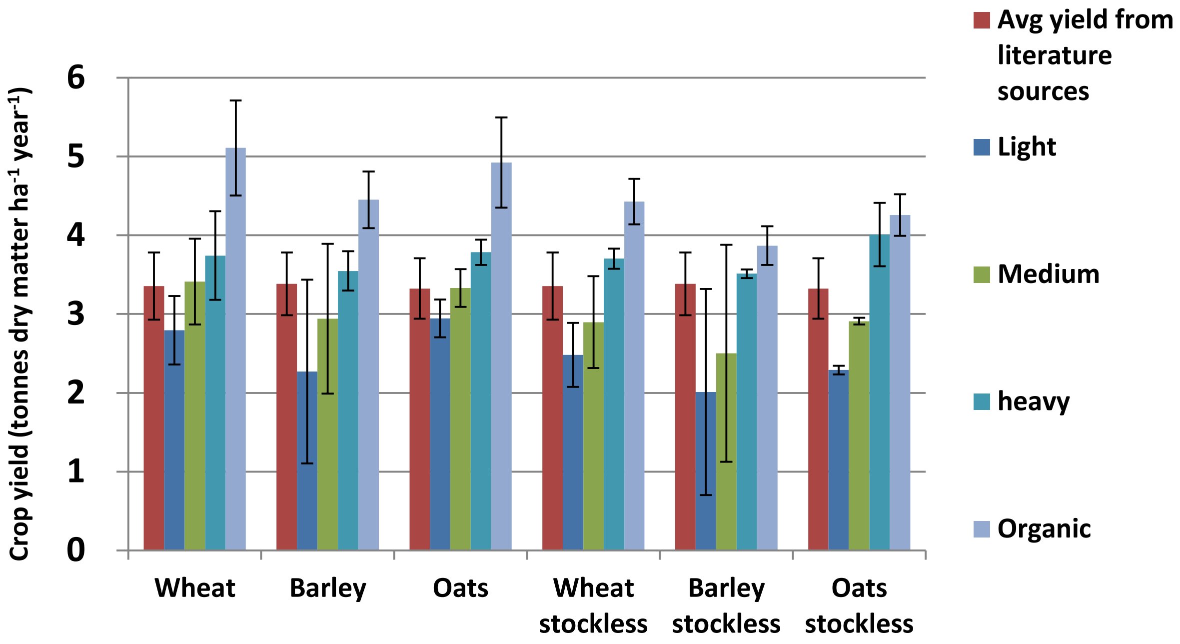

Figure B.3: Adjusted yield for organic crops grown on a range of soil types (avg. N fix scenario only) error bars = s.d. Organic = high organic matter soil (i.e. >15% organic matter)

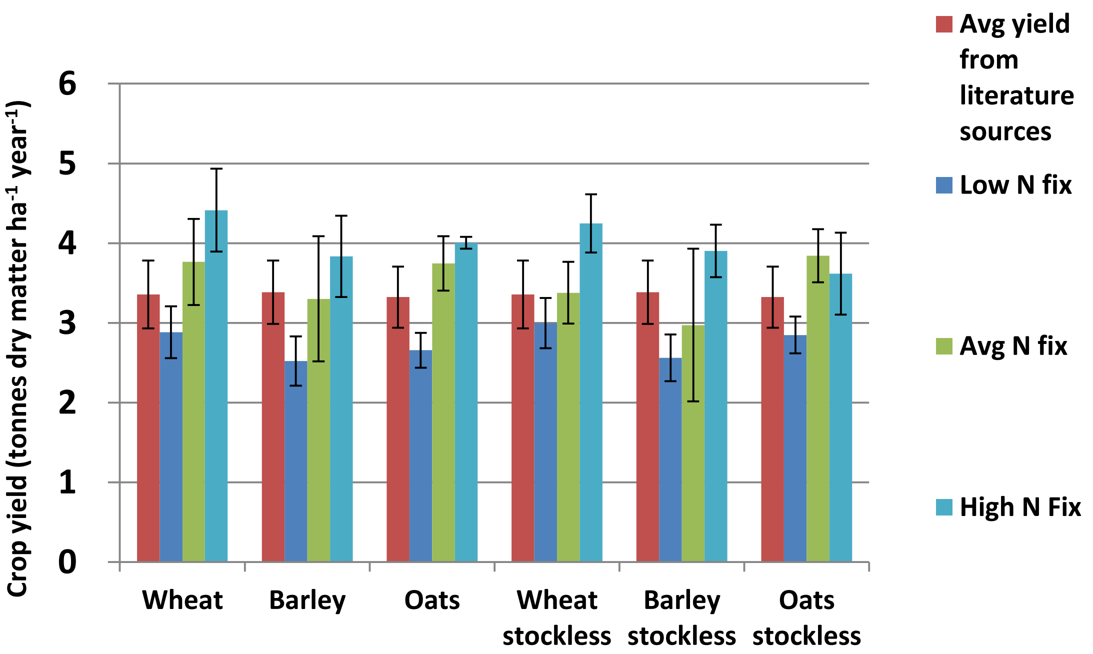

Figure B.4: Adjusted yield for organic crops by N fixation rate within NDICEA. Error bars = s.d.

For all crops higher yields were found under dry conditions and on heavy and organic soil types. Adjusting the N fixation rate had the greatest effect on potential yield (Figure B.4). For all rotations total product removal was significantly higher on dry soils in relation to wet and very wet conditions ($P<0.01$). Product removal was also significantly higher for heavy and organic soils compared to light and medium soil types respectively ($P<0.05$).

B.3.1 Pasture yields

As the focus of NDICEA is on rotational systems (Van der Burgt *et al.*, 2006) a separate model was applied to estimate pasture yields. Linear regression equations were calculated based on the relationship between grassland yield and site class using data reported in Brockman (1994), see Figure B.5:

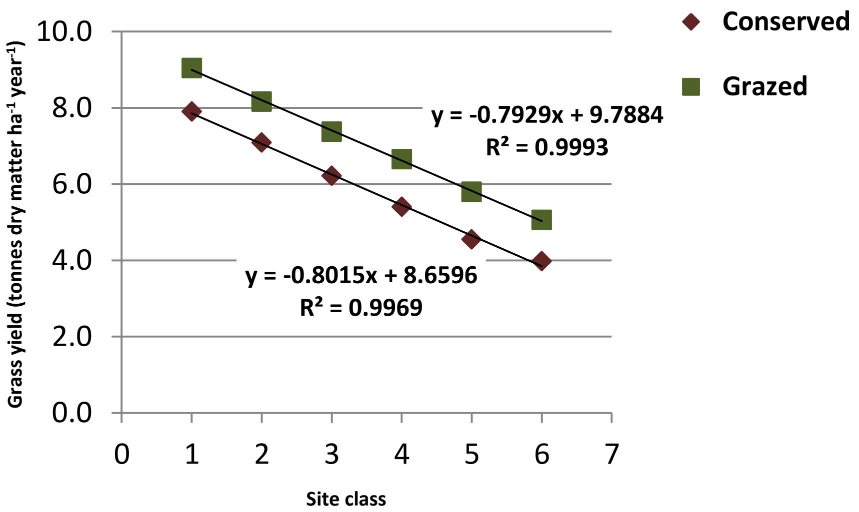

Figure B.5: Linear regression equations defined for conserved/ensiled and grazed grassland outputs by site class, based on yields and class descriptions in Brockman (1994)

The regression model was applied for a range of site classes, with each class defined by soil type, altitude and annual rainfall using a method derived from Williams *et al.* (2006):

$$\text{Site class } \omega = S + \beta + \gamma \tag{1}$$

Where:

$S =$ 1 for heavy soils, 2 for medium soils, 3 for light soils

$\beta =$ 2 if altitude >300m, $\beta = 1$ if altitude >100m, else zero

And:

$\gamma = 0.225 + 1.95\,(z/100) - 0.5\,(z/100)^2$

$z = 0.4R$ where R = annual rainfall

For the purpose of this study the altitude categories >300m and >100 were used to represent Severely Disadvantaged Areas (SDA) and Less Favoured Area (LFA) respectively.

Site classes were produced for the grassland-dominated organic conversion trial at the University of Wales, Aberystwyth (Haggar and Padel, 1996) and for the long-term organic systems trial at Scotland's Rural College (SRUC,Taylor et al., 2006) using the method described above. Grassland yield estimates were produced using the regression model described in Figure B.5 and compared to the recorded values from both experimental sites (Figure B.6 and Figure B.7):

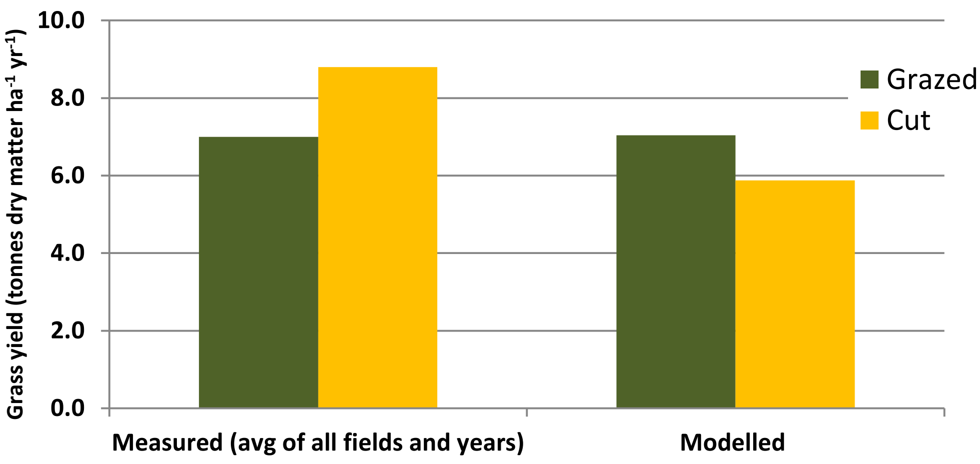

Figure B.6: Comparison of modelled permanent grassland outputs and recorded yields from Ty Gwyn Organic Dairy Conversion project (Haggar and Padel (1996)

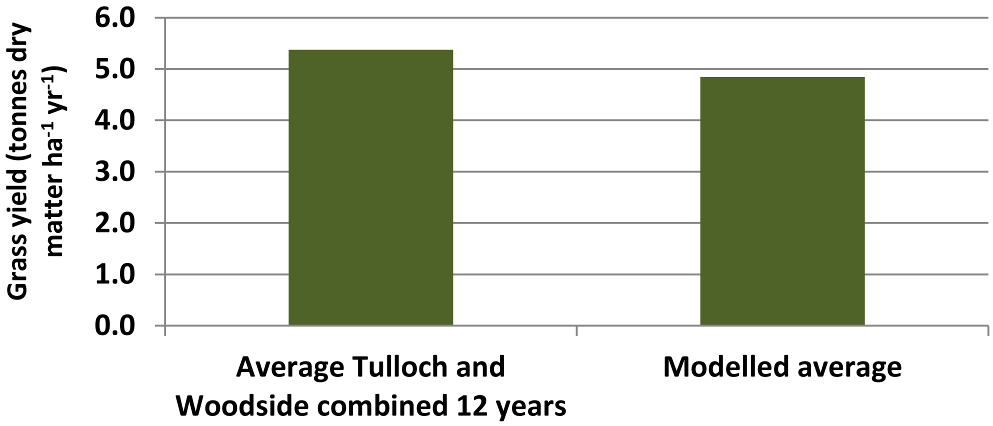

Figure B.7: Comparison of modelled grassland outputs and average cut/grazed yields from SRUC (Taylor *et al.* 2006)

Modelled outputs were close to the recorded values at both sites (i.e. within 1-2 tonnes-DM ha^{-1} yr^{-1}). Grassland yields were therefore estimated for each of the 16 land classes described in Figure 1. Outputs for each class (tonnes of DM $ha^{-1}yr^{-1}$) are displayed in Table B.4 below:

Table B.4: Modelled pasture yields for each land class and altitude (>300m = SDA, >100m = LFA). Average yield of grazed and conserved grass (tonnes dry matter ha^{-1} yr^{-1}). Organic = high organic matter soil (i.e. >15% organic matter)

	Non LFA			
Rainfall/soil type	Organic	Heavy	Medium	Light
dry	6.0	6.8	6.0	5.2
medium	6.2	7.0	6.2	5.4
wet	6.5	7.3	6.5	5.7
very wet	8.0	8.8	8.0	7.2
	LFA			
Rainfall/soil type	Organic	Heavy	Medium	Light
dry	5.2	6.0	5.2	4.4
medium	5.4	6.2	5.4	4.6
wet	5.7	6.5	5.7	4.9
very wet	7.2	8.0	7.2	6.4
	SDA			
Rainfall/soil type	Organic	Heavy	Medium	Light
dry	3.2	5.2	5.1	5.0
medium	3.4	5.2	5.1	5.1
wet	3.7	5.3	5.2	5.1
Very wet	5.3	5.4	5.3	5.3

B.4 References

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Appendix C Optimal Land Use Model (OLUM) description

A Linear Programming (LP) approach was selected to assess the production and land use impacts of a 100% organic England and Wales. LP is an optimisation method used for maximising an objective subject to other constraints (Jones and Salter, 2013). LP approaches have been widely applied for the assessment of policy and/or changes in environmental factors (e.g. CO_2 concentrations and temperature, Audsley *et al.*, 2006).

The programming language GAMS (General Algebraic Modelling System, www.gams.com) was selected for the construction of the model. GAMS offers several advantages over other common languages. In particular by allowing for simple model classification through the use of mathematical symbols and algebraic relationships it is possible to define models in unambiguous terms that can be readily understood and adjusted. In addition the language includes in-built features for error-detection and has compatibility with Excel through a GAMS Data Exchange (GDX) facility. All identifiers used within the GAMS code must also be declared and described with associated text before being referenced in the model, which can be of great assistance when returning to models after a period of absence in addition to helping with understanding of the modelled processes and interactions. The large library of existing models and an online GAMS community (http://www.gamsworld.org) allows for sharing of ideas and approaches between those facing similar challenges (Bussieck and Meeraus, 2004). For more information on the language and examples of its application see http://www.gams.com/.

C.1 Model overview and objective function

The Optimal Land Use Model (OLUM) developed for this study takes a biophysical approach to modelling production and land use under an organic scenario. The primary objective function is to maximise the energy output (expressed in metabolisable energy - ME) of farming in England and Wales, i.e.

$$\textbf{Maximise}: \mathbf{Z} = \sum_{ij=0}^{n} \boldsymbol{C_{ij}} \cdot \boldsymbol{x_{ij}} \tag{1}$$

$$\textbf{subject to} \quad \boldsymbol{R}\mathbf{x}(\mathbf{ij}) \leq \mathbf{b};\ \mathbf{x}(\mathbf{ij}) \geq \mathbf{0}$$

Where Z represents the objective variable (i.e. total food output as ME), C_{ij} is the energy output in of individual organic agricultural products (i) on each soil and rainfall class (j) and x_{ij} is activity levels (land areas for crops and livestock numbers). Rx_{ij} represents the resource requirements of producing crops or livestock (x_{ij}) and b is the resource endowment and input availability vector (e.g. manure-N, land by soil and rainfall class). Constraints were specified as linear inequalities and equalities (e.g. $A \cdot x < b$, $A \cdot x = b$) determined in accordance with:

i. Availability of land by farm type and land class
ii. Maximum/minimum crop areas (ha) by crop groups (i.e. rotation constraints)
iii. N supply limits (including crop and livestock offtake, atmospheric deposition and biological nitrogen fixation by legumes)
iv. Maximum and minimum stocking densities (livestock units per ha)
v. Livestock numbers, with young-stock (e.g. store cattle, finishers) defined as a proportion of breeding stock (e.g. suckler cows, dairy cows)
vi. Annual feed requirements by livestock feed type in metabolisable energy and crude protein (CP)
vii. Limits on the total production (tonnes) of individual crops and livestock products, set at 150% of the current supply on the assumption that further increases could not be absorbed by the market.

C.2 Land availability constraints

Land availability by farm type and land class were determined using data from the 2010 June Agricultural Census (JAC) combined with a National Soil Resources Institute (NSRI) dataset and 30-year rainfall data from the UK Met Office. Land areas for market gardens, orchards and field vegetables were also derived from the

JAC and treated as separate farm-types within the model. Total land availability limits for each farm-type and land class were defined as shown below:

$$\sum_{c=0}^{n} a_{c,t,s,r} = L_{t,s,r} \quad \forall\ t,s,r \tag{2}$$

Where $a_{c,t,s,r}$ is total crop area and $L_{t,s,r}$ land availability by farm type (t), land class (s) and NUTS1 region (r). An assumption is made that the area of each robust farm type will remain the same under organic management as evidence suggests that for the vast majority of farms, the dominant enterprise will remain in place post-conversion to organic management (Howlett *et al.*, 2002; Langer, 2002).

The yield by soil and rainfall class was set as a parameter within the model, drawing on the results from the modelling with NDICEA and the coverage of each soil/rain class (see Appendix B).

C.3 Maximum/minimum crop area constraints

Limits on crop areas were defined by crop-area constraints constructed for each farm type, using data from three sources providing technical information to the organic sector (i.e. Davies and Lennartsson, 2005; Lampkin, 2002; Lampkin *et al.*, 2014) and crop areas reported within a three year matched sample of organic farms in England and Wales (Moakes *et al.*, 2012; 2014, see Table 7 below). Constraints (i.e. maximum crop areas) were defined by crop group as illustrated in equation 3:

$$\sum_{g=0}^{n} a_{g,t,s,r} \geq L_{t,s,r} \cdot R_{g,t} \quad \forall\ t,s,r \tag{3}$$

Where $a_{g,t,s,r}$ is total land-area by crop group (g), $L_{t,s,r}$ is land area within each farm type, soil/rainfall class and region (t, s, r) and $R_{g,t}$ is the coefficient of land area (i.e. a proportion of utilisable agricultural area - UAA). For the purpose of this study crop groups were based on growth characteristics (e.g. cereals, beans and peas, alliums, root vegetables - see Appendix B). Crops were also defined as "stocked" or "stockless" within each group and by their end-use (i.e. livestock feed or human consumption). Maximum crop-group areas were defined with the same equation structure, using a "less than or equal to" sign (i.e. $\leq$ in place of the $\geq$ shown above).

C.4 Nitrogen supply and offtake constraints

Nitrogen supply and crop/livestock offtake was defined on a regional basis, to allow for transfer of manure between farms within the same area, as shown in equation 4:

$$\sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} co_{c,t,s,r} \cdot a_{ch,t,s,r} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} livn_l \cdot l_{l,t,s,r}$$
$$\leq \sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} fx_{c,t,s,r} \cdot a_{c,t,s,r} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} lu_{l,t,s,r} \cdot l_{l,t,s,r} \cdot Nin_t$$
$$+ \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} imp_{ltsr} \cdot compn + \sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} a_{c,t,s,r} \cdot dp_{c,t,s,r} \quad \forall\, r \tag{4}$$

Where $co_{c,t,s,r}$ is N offtake per ha by crop type, $a_{ch,t,s,r}$ is the area of crops destined for human consumption, $livn_l$ is the livestock N-offtake per head of livestock $l_{l,t,s,r}$. The term $fx_{c,t,s,r}$ represents N fixation per hectare of crop $a_{c,t,s,r}$, $lu_{l,t,s,r}$ is livestock units, $l_{l,t,s,r}$ is livestock numbers and Nin_t is N contained within imported concentrate (i.e. cereals and beans). Imported compound feed (e.g. soy cake) is represented by $imp_{l,t,s,r}$ and $compn$ which represents N content/tonne, based on feed values within Watson et al. (2010). The term $dp_{c,t,s,r}$ represents average atmospheric nitrogen deposition values, derived from national pollution data downloaded from the Centre for Ecology and Hydrology (CEH) website (http://www.pollutantdeposition.ceh.ac.uk/). N supply and offtake values for crops and livestock products were derived from Defra Fertiliser Recommendations (Defra, 2010) and the nutrient budgeting software PLANET (version 2.2).

To capture manure requirements for individual crops, a separate constraint was applied within each region, as shown in equation 5:

$$\sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} cropm_{c,t,s,r} \cdot a_{c,t,s,r} \leq \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} livm_{l} \cdot l_{l,t,s,r} \qquad \forall\, r \tag{5}$$

Where $cropm_{c,t,s,r}$ represents manure-N requirements per hectare and $livm_l$ total manure N produced over the housed period by livestock (l) within each region (r). The amount of manure-N supplied by each livestock type was derived from reference figures for organic farming inspections (Schmidt *et al.*, 2009) in addition to data derived from guidance on managing manure on organic farms, and information on Nitrogen Vulnerable Zones (NVZ) management in the UK (Defra, 2009). Manure application limits per hectare were also defined, in accordance with organic standards which set a maximum application rate of 170 kg-N per hectare of UAA (Council Regulation No 889/2008, 2008):

(6)

$$\sum_{c=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} a_{c,t,s,r} \cdot 170 \geq \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n} l_{l,t,s,r} \cdot livman_{l} \quad \forall\, r$$

Where $a_{c,t,s,r}$ is crop area, $l_{l,t,s,r}$ is livestock numbers and $lvman_l$ is the total kg of manure-N applied within each region (r).

C.5 Stocking constraints

Livestock numbers were derived for each robust farm type using a 3-year average of Farm Business Survey data reported in Moakes et al. (2012, 2014). Minimum stocking rates were defined as shown in equation 7:

$$\sum_{l=1}^{n} lu_l \cdot l_{l,t,s,r} \geq \sum_{c=1}^{n} c_{c,t,s,r} \cdot srmin_t \quad \forall\; t,s,r \tag{7}$$

Where lu_l represents livestock units, $l_{l,t,s,r}$ is livestock numbers, $c_{c,t,s,r}$ the cropping area and $srmin_t$ the minimum stocking rate within each farm type, soil/rain class and region (t,s,r). Minimum stocking rate constraints were removed for cereal, field vegetable, market garden and general cropping farms to allow for stockless production. Maximum stocking rates were defined using the same equation structure, as shown in equation 8, where $srmax_t$ is the maximum rate per ha:

$$\sum_{l=1}^{n} lu_l \cdot l_{l,t,s,r} \leq \sum_{c=1}^{n} c_{c,t,s,r} \cdot srmax_t \quad \forall\; t,s,r \tag{8}$$

Livestock young/rearing stock numbers and replacements were calculated as a proportion of the dominant livestock type (the stock type with the highest number of livestock units as a proportion of the total livestock, e.g. dairy and suckler cows on dairy farms and lowland grazing farms respectively). An example of this approach is shown in equation 9:

$$\sum_{bc=1}^{n}\sum_{lg=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} l_{bc,lg,s,r} \cdot pl_{fc} = l_{fc,s,r} \quad \forall \;\; l, lg, s, r \tag{9}$$

Where $l_{bc,lg,s,r}$ is the dominant livestock type (in this example beef suckler cattle, bc) pl_{fc} is a fixed proportion defined for store cattle. The term $l_{fc,s,r}$represents total store cattle numbers and lg is the area of the farm-type "lowland-grazing" in each soil/rain class (s) and region (r).

C.6 Feed supply constraints

Total feed supply limits were defined using the following feed-groups:

- Forage (e.g. grass/clover, fodder beet, fodder maize)
- Concentrates (e.g. cereals, beans, peas)
- Compound feeds (e.g. soybeans, crop processing residues, other imported feed)

The proportion of total livestock ME demand supplied by each group was calculated for each robust farm type, using data reported in Moakes *et al.* (2012, 2014, see Table C.3). Based on these proportions, upper limits were specified for each region and robust farm type, as shown in equation 10 below:

$$\sum_{rum=1}^{n}\sum_{s=1}^{n} l_{rum,t,s,r} \cdot req_{rum} \cdot frg_{rum} \leq \sum_{cf=1}^{n}\sum_{s=1}^{n} c_{cf,t,s,r} \cdot cropen_{cf,s} \quad \forall \; t,r \tag{10}$$

Where *rum* is ruminant livestock (a subset of livestock, l), req_{rum} is the total ME requirement, frg_{rum} is the proportion of ruminant livestock ME requirement provided by forage, $c_{cf,t,s,r}$ is crop area for forage crops, cf (expressed in hectares) encompassing pasture, temporary grassland and other forage (e.g. fodder beet) and $cropen_{cf,s}$ is the crop energy yield (in ME) for forage crops defined by soil and rainfall class (*s*).

Unlike forage, concentrate and compound feed supply/usage was defined at national level, as shown in equation 11:

$$\sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} l_{l,t,s,r} \cdot req_{l} \cdot conc_{t} \leq \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} a_{cc,t,s,r} \cdot cropen_{cc,s} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} comp_{l,t,s,r} \cdot compMJ \tag{11}$$

Where $l_{l,t,s,r}$ is livestock numbers req_{l} is the ME requirement and $conc_{t}$ is the proportion of the feed demand within each farm type *t* met by concentrate and/or compound feed. The term $a_{cc,t,s,r}$ represents concentrate feed crop (cc) area and $cropen_{cc,s}$ the ME supply per hectare on each soil and rainfall class. The supply and energy-content for compound feed is represented by $comp_{l,t,s,r}$ and the scalar *compMJ*. Protein requirements and supply were defined using a similar approach, as shown in equation 12:

$$\sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} l_{l,t,s,r} \cdot protreq_l \leq \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} a_{lc,t,s,r} \cdot croprot_{cc,s} + \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} comp_{l,t,s,r} \cdot comprot \tag{12}$$

Where $protreq_l$ is the crude protein demand for livestock ($l_{l,t,s,r}$), $a_{lc,t,s,r}$ represents livestock feed crop area and $croprot_{cc,s}$ the crude protein yield of cereals and beans for livestock feed by soil and rainfall class (s). The supply and crude protein content for compound feed is represented by *comp* and the scalar *comprot.*

Maximum contributions of concentrate and compound (i.e. imported) feed types were imposed to ensure an adequate feed composition and avoid harmful nutritional effects (e.g. from over-feeding of beans which have high tannin contents (Mueller-Harvey, 2006) as shown in equation 13:

$$\sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} l_{l,t,s,r} \cdot req_l \cdot maxbean_t \leq \sum_{l=1}^{n}\sum_{t=1}^{n}\sum_{s=1}^{n}\sum_{r=1}^{n} a_{bp,t,s,r} \cdot cropen_{bp,s} \tag{13}$$

Where $maxbean_t$ is the maximum proportion of ME supplied by beans and/or peas (*bp*) within each farm type *t*. The same equation structure was applied for cereal crops.

Compound feed was defined as the sum total of soya, cereal and other crop processing residues, and other feed as shown in equation 14 below:

$$\sum_{l=1}^{n}\sum_{s=1}^{n} comp_{l,t,s,r} = \sum_{l=1}^{n}\sum_{s=1}^{n} soya_{l,t,s,r} + \sum_{l=1}^{n}\sum_{s=1}^{n} residue_{l,t,s,r} + \sum_{l=1}^{n}\sum_{s=1}^{n} other_{l,t,s,r} \quad \forall \; t,r \tag{14}$$

where $comp_{l,t,s,r}$ is total compound feed (weight in tonnes). Upper limits on compound feed rates were imposed by farm type and region, as shown in equation 15:

$$\sum_{l=1}^{n}\sum_{s=1}^{n} comp_{l,t,s,r} \cdot compMJ \leq \sum_{l=1}^{n}\sum_{s=1}^{n} l_{l,t,s,r} \cdot req_l \cdot complim_t \quad \forall \; t,r \tag{15}$$

Where $comp_{l,t,s,r}$ and $compMJ$ is the tonnage of compound feed and the energy content expressed as ME, req_l is the ME requirement of livestock and $complim_t$ is the proportion of ME supplied by compounds within farm type t and region r. To ensure an adequate supply of amino acids to monogastric livestock (pigs and poultry) within compound feed, soya requirements were defined based on composition data supplied by Vitrition Organic Feeds, Edwards *et al.* (2002) and Jones *et al.* (2014), as shown in equation 16 below:

$$\sum_{l=1}^{n}\sum_{s=1}^{n} l_{l,t,s,r} \cdot req_l \cdot soyr_t = \sum_{l=1}^{n}\sum_{s=1}^{n} soyl_{l,t,s,r} \cdot smj \quad \forall \; t,r \tag{16}$$

Where req_l is the sum of the total ME requirement of livestock ($l_{l,t,s,r}$) within each farm type *t* and region, $soyr_t$ is the total proportion of the ME demand supplied by soya meal. $soyl_{l,t,s,r}$ and smj are the total weight of soya meal and the energy density respectively. The crop processing residue-component of compound feed was also defined as a fixed proportion, based on the tonnes of oilseed rape, sugar beet, milling wheat and malting barley produced in 2010 (Defra, 2015) and tonnes of product recovered as wheatfeed, oilseed rape cake, sugar beet pulp and distillery by-products respectively (Defra Feedingstuffs Survey, 2016).

Crop processing residue-feed was calculated as a fixed proportion based on the tonnes of oilseed rape, sugar beet, milling wheat and malting barley produced in 2010 and tonnes of product recovered as wheatfeed, oilseed rape cake, sugar beet pulp and distillery by-products respectively, based on data within the Defra Compound Feedingstuffs Survey (2016):

Table C.1: tonnes of crop-residue feed produced in UK as % of export-adjusted production:

By-product feedstuffs	000' Tonnes produced in UK - 2010	% of production - 2010:
Wheatfeed	852	13%
Oilseed rape cake and meal	771	36%
Dried sugar beet pulp	274	4%
Barley distillery by products	317	16%

An overview of the OLUM and the interactions between components is presented in Figure C.1 below:

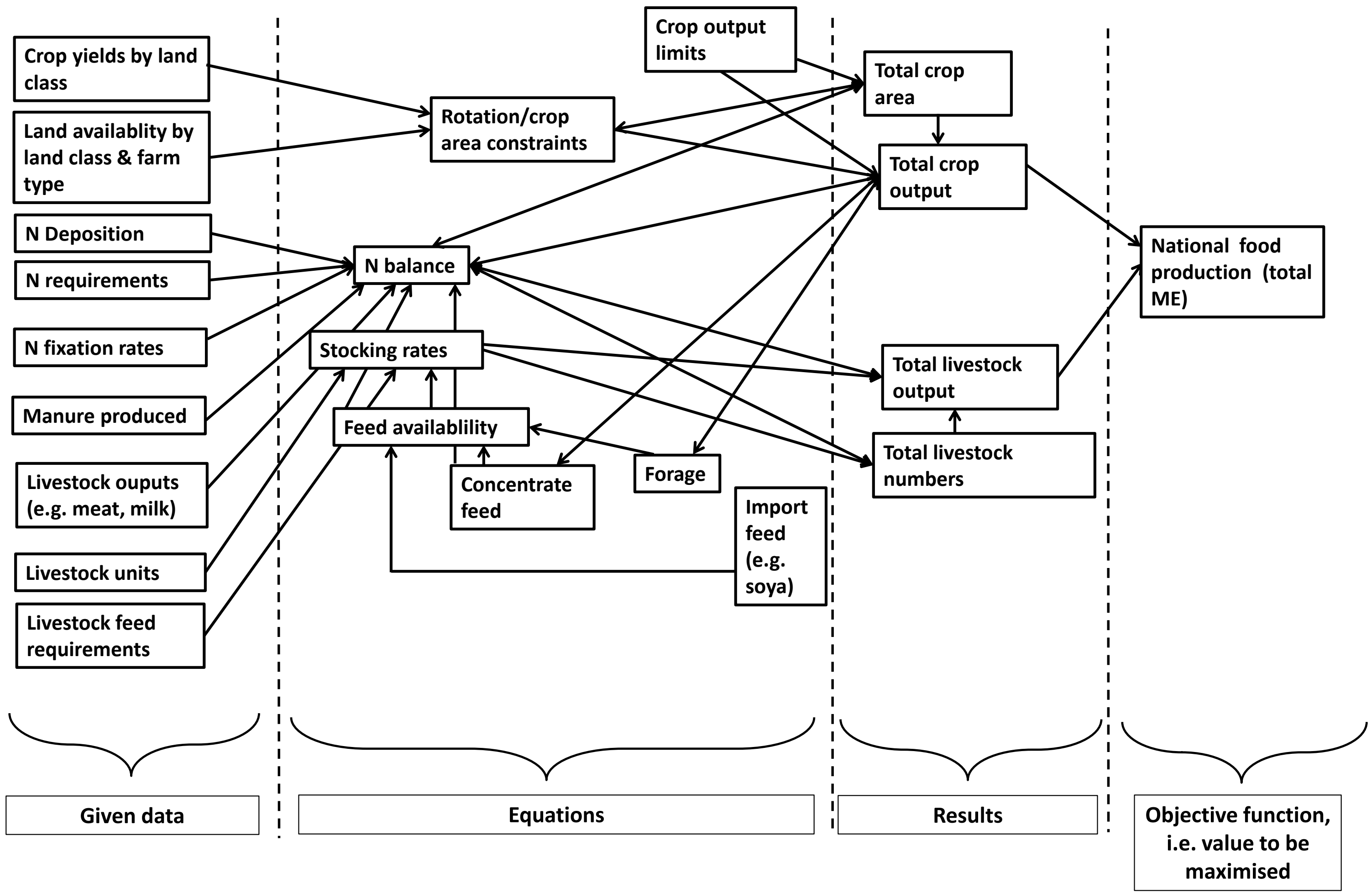

Figure C.1: overview of the OLUM and interactions between components

C.7 Example input data for Lowland Grazing Robust Farm Type

Within the OLUM typical organic farm structures informed the development of constraints (e.g. with regard to stocking rates, livestock feed and crop areas). Farm structures were defined from a three-year matched sample reported in Moakes et al. (2012, 2014) with average yields used to ascertain feed-energy outputs at three rates of N-fixation (see Appendix B). The total feed ME supplied by each farm-type was used to determine the need for import of concentrate and bulk feed requirements. In cases where there was an excess (e.g. cropping farms with a low stocking rate) cereal and/or legume crops could be exported for human consumption and/or to other farms for feed supply. An example of a typical farm structure is presented in Table C.2.

Table C.2: Crop areas and outputs for Lowland Grazing Farm Type

	2009/10	2010/11	2011/12	Average	*Std. dev*	Yield low	Yield medium	Yield high
Sample number	26	26	34	28.7	*3.8*			
Average farm size (ha)	107	107	138	117.3	*14.6*			
Land use (ha)								
Wheat	0	0	3.2	1.1	*1.5*	2.7	4.1	4.8
Barley	0.4	0.6	2.4	1.1	*0.9*	2.6	3.6	4.5
Other cereals	1.1	0.9	1.8	1.3	*0.4*	2.7	3.6	4.3
OSR						0.8	1.2	1.5
Peas/Beans	0	0	0.8	0.3	*0.4*	2.3	4.1	5.0
Potatoes						16.0	25.0	34.7
Sugarbeet						22.6	39.7	43.3
Horticulture						9.6	15.0	20.8
Other crops						1.1	1.4	1.6
Tillage - fodder	3.3	2.9	3.9	3.4	*0.4*	61.5	92.9	116.4
Grassland (grazing, hay, silage	101	101.5	119.1	107.2	*8.4*			
Permanent grass	79.0	71.9	80.2	77.0	*3.7*	2.0	5.6	7.2
Temporary grass	22.0	29.6	38.9	30.2	*6.9*	4.5	10.6	13.8
Fallow and land let	1.3	1.4	3.9	2.2	*1.2*	0.0	0.0	0.0
Rough grazing	0.2	0.2	2.8	1.1	*1.2*	1.4	5.3	7.0
Total area (UAA)	107.3	107.5	137.9	117.6	*14.4*			
Total tillage area	26.8	34.0	51.0	37.3	*10.1*			

Livestock metabolisable energy (ME) requirements were drawn from industry data sources (Lampkin *et al.*, 2011; The Professional Nutrient Management Group, 2015) and multiplied by the total number of each livestock type, as shown in Table C.3.

Table C.3: Livestock numbers and total feed requirements

Livestock numbers	2009/10	2010/11	2011/12	Average	*Std. dev*	ME requirement	Total ME required based on livestock numbers
Beef cows	33	33	37	34.3	*1.9*	29,500	1,012,833
Calves	36	36	55	42.3	*9.0*	6,750	285,556
Stores	17	17	25	19.8	*3.8*	13,600	269,490
Finishers	24	20	19	20.8	*2.2*	27,050	562,717
Bulls	2	2	2	1.7	*0.3*	32,321	56,360
Breeding sheep	139	154	246	179.7	*47.3*	6,375	1,145,375
Other sheep	123	129	228	160.0	*48.1*	3,000	480,000
Poultry	14	0	85	33.0	*37.2*	554	18,282
Other livestock	1	1	1	1.0	*0.0*	29,061	29,061

Feed ME supply by source was defined through official sources on feed rates (AHDB Dairy, 2012; Lampkin *et al.*, 2011) and the total amount of feed-ME available, based on crop areas and outputs. Where concentrate fed on farm exceeded the amount grown, the additional concentrate (i.e. the additional ME) was imported. Imported bulk feed was defined separately to concentrate, using composition data from Vitrition Organic Feeds, Jones *et al.* (2014) and Newcastle University (Edwards, 2002).The proportions of feed by type were applied within the feed composition constraints described above to avoid over-supply of certain feedstuffs with ant-nutritive features (e.g. beans and peas) and to ensure adequate supply of essential amino acids in the case of pigs and poultry.

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Appendix D Eatwell Food Group allocation

The Eatwell food groups are based on recommendation for a well-balanced diet and provide a structure for the development of healthy eating guidelines. Within this study crops and livestock products were allocated to each group to compare outputs at regional and national scales (Table D.1). The outputs of food as metabolisable energy (ME) under a 100% conversion to organic agriculture are also shown for each of the Eatwell groups (Figure D.1).

Table D.1: Eatwell groups and crop / livestock product allocation

Fruit and veg	**Starchy Carbohydrates**	**Non-dairy protein**	**Milk and dairy**	**Sugary foods**
Onions and leeks Beetroots Carrots Broccoli Cabbage Lettuce Squash Parsnips Turnips Tomatoes Cucumbers Peppers Protected salad Apples Pears Strawberries Other combined hort	Potato Wheat Oats Rye Barley	Red Meat Eggs Broad beans Runner beans Green peas Dry peas Poultry meat	Milk	Sugar beet

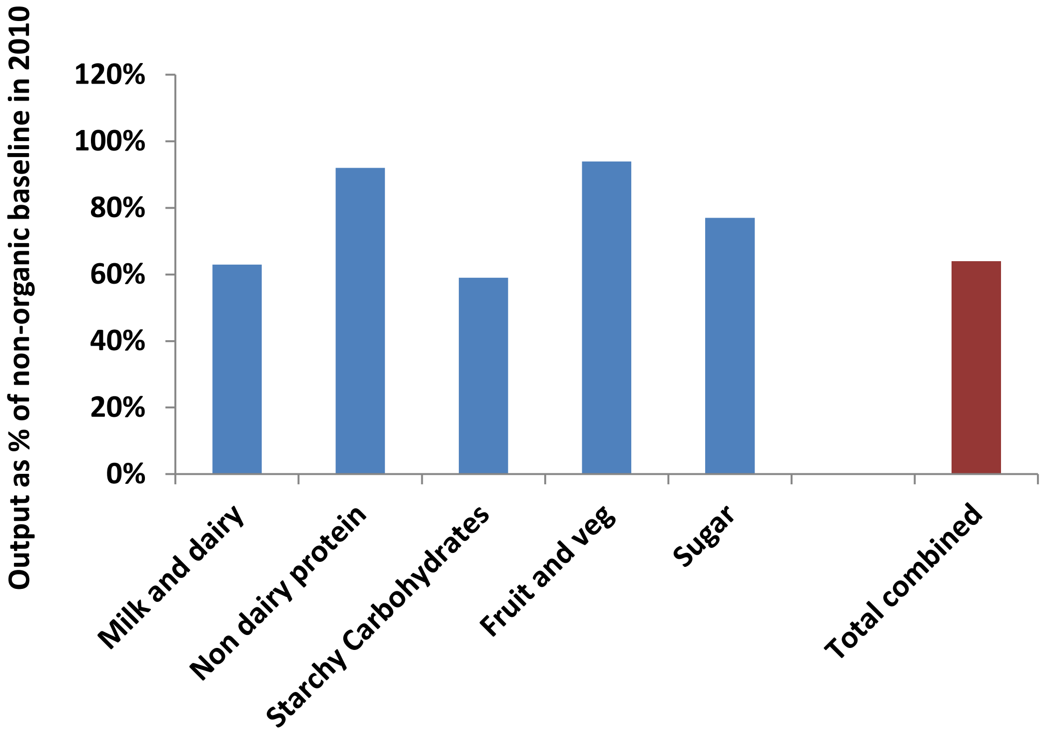

Figure D.1: production of food as metabolisable energy (ME) for each Eatwell group under a 100% conversion to organic agriculture in England and Wales

Appendix E Data sources used for Life Cycle Assessment study

E.1 Organic systems data

A range of industry datasets were applied within the Optimal Land Use Model (OLUM) and the Agri-LCA models to represent technical and biological conditions within organic agriculture. The data sources applied are summarised in Table E.1.

Table E.1: data sources used to represent organic production systems

Information extracted	Data source	Reference
Crops		
Organic crop yields	Ecoinvent database Organic Farm Management Hanbook European Commission project report	Frischknecht et al. (2005) Lampkin et al. (2012) Audsley et al. (1997)
Manure and compost application rates	Defra project on environmental burders of agriculture	Williams et al. (2006)
Typical cultivation practices by crop type	Defra project on environmental burders of agriculture	Williams et al. (2006)
Livestock		
Replacement rates Average milk yield/annum Concentrate feed rates	AHDB Dairy 3 year Carbon Footprinting Study	AHDB Dairy (2014)
Mortality rates Liveweights / finsihing ages	Organic Farm Management Handbook	Lampkin et al. (2012)
Mortality rates Eggs / year Stocking rate Birds / ha	LCA studies of organic poultry	Leinonen et al. (2012a,b)
Poultry deadweight	Organic poultry carcass trait study	Rizzi et al. (2007)
Poultry feed composition	Virition organic feeds	Vitrition organic feeds (2015)
Concentrate feed composition	Reading University Study	Jones et al. (2014)
Pig diet composition	Newcastle University Organic Pig rearing guide	Edwards et al. (2002)

In addition organic farm structure data (i.e. land areas by crop type and stocking rates) were drawn from a panel of organic farms within the Farm Business Survey for England and Wales (i.e. values for the years 2009/10 – 2011/12 by Robust Farm Type, as reported by Moakes *et al.* (2012, 2014).

E.2 Non-organic systems data and imports/exports

Data used for the non-organic comparison, and for the calculation of imports and exports to England and Wales are described in Table E.2.

Table E.2: Non-organic production and import / export data sources

Information extracted	Data source	Reference
Crops		
Arable crop production volumes and imports / exports	Agriculture in the United Kingdom	Defra (2015a)
Sugar production data	FAO Sugar beet handbook	FAO (2009)
Vegetable and fruit production volumes and imports / exports	Basic Horticultural Statistics 2013	Defra (2013)
Livestock		
Pork production, imports / exports	UK annual numbers of livestock slaughtered by Country	Defra (2015b)
Pork production, imports / exports	The BPEX Yearbook 2014-2015	AHDB Pork (2015a)
Beef and lamb production volumes by country	AHDB Beef Cattle and Sheep yearbook 2013	AHDB Beef & Lamb (2013a, b)
Beef and lamb consumption and imports / exports	Hybu Cig Cymru Little Book of Meat Facts	Hybu Cig Cymru (2012)
Poultry meat production volumes by country	E&W Poultry Slaughterhouse Survey	Defra (2015b)
Poultry meat and eggs, imports / exports	AHDB Poultry Pocketbook	AHDB Pork (2015b)
Livestock numbers	June Census 2010	Defra (2011)
Eggs produced / hen	LCA of egg production systems in the UK	Leinonen et al. (2012a, b)
Milk production volume, imports and exports - England and Wales	Defra Milk Utilisation Data	Defra (2016)

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