Life Cycle Assessment of UK Pig Production Systems:

The impact of dietary protein source

Katie Louise Stephen

Master of Philosophy
The University of Edinburgh
2012

Declaration

I declare that this thesis is my own composition. I wrote the manuscript and the work presented was conducted by me. The work has not been submitted for any other degree or professional qualification.

Katie Louise Stephen
April 2012

Acknowledgements

This two-year studentship project was sponsored by BPEX, and I would like to thank them for this opportunity. I would also like to sincerely thank my (former) SAC supervisors for all of their support, i.e. Dr Bert Tolkamp, Dr Kairsty Topp, Dr Jos Houdijk and Prof Ilias Kyriazakis. In addition, I would like to thank the following people and organizations who helped in sourcing crucial data for the project described in this thesis. Alison Johnson of BQP kindly provided typical diet specifications as used in the pig feeding industry, Prof Alvaro Da Silva provided advice and knowledge on Brazilian soya production, BADC provided the necessary UK weather data, Robin Walker for his time discussing various aspects of existing LCAs, Elaine Booth and Lawrence Morrice for their advice on UK grown legumes and Ian Bingham for his advice on crop rotations.

This project was aligned to the larger Defra-LINK project LK0682 (Green Pig). Its academic partners are SAC, University of Nottingham and NIAB, and it is financial supported by BOCM Pauls, BPEX, Evonik-Degussa, Genesis QA, Harbro, Premier Nutrition Products, PGRO, QMS, Soil Association and UNIP, with Defra match funding through the Sustainable Livestock Production LINK program.

I would also like to thank my good friends and colleagues at SAC who have provided great support, motivation and advice. Especially Louise Buckley and Ramona Donald for their company and good humour on late night working sessions, and for the endless supply of coffee. Also, Steven Varker for his moral support and putting up with my constant talk of ‘the thesis’. My parents and Marilyn Morris for their enthusiasm and always believing I could do it.

Abstract

A Life Cycle Assessment (LCA) was developed to evaluate the environmental impacts of producing 1 kg pig live weight. A comparison was made between dietary protein sources, i.e. imported soybean meal with the UK protein sources (1) peas, (2) beans and (3) lupins. A holistic approach was used and the LCA was developed using several sub-models to include all processes within the system boundaries for pigs grown from 12 kg to 106 kg. Two UK sites were modelled, East Anglia and Yorkshire, each with individual site conditions and a comparison of the two sites was included using a common soil type present at both sites. A Brazilian corn-soya rotation was simulated for the production of soybean meal. Individual soil and climate conditions were defined at each site and two fertilizer scenarios were modelled: synthetic and slurry. The environmental impacts assessed were (1) Global Warming Potential (GWP), (2) Eutrophication and (3) Acidification. Differences occurred between diet and sites but also between fertilizer scenarios. It was concluded that the GWP per kg pig in the slurry fertilizer scenarios are consistently higher. The bean based diets resulted in the lowest GWP ranging from 1.85 to 2.67 kg CO_2 equivalent^{100} and the soya based diets with the highest GWP per kg pig, 2.52 to 3.08 kg CO_2 equivalent^{100}. Diet production contributed the most to GWP per kg pig, i.e. 63.9 – 78.5 %. Transport contributed approximately 1% to GWP in the home grown diet scenarios, however in the soya based diet scenarios, this was on average 3 %. Eutrophication potentials were higher in the synthetic fertilizer scenario. The lupin based diets were associated with the highest eutrophication potential, 0.056 – 0.133 kg PO_4 equivalent in both fertilizer scenarios. Whereas the pea based diets were consistently associated with the lowest eutrophication potential, 0.049 to 0.103 kg PO_4 equivalent. The soya based diets therefore concluded with the highest acidification potential, 0.054 to 0.129 kg SO_2 equivalent in both fertilizer scenarios. The results were weighted from the lowest to highest results for each environmental impact category for each diet scenario at each site. The overall conclusion is that the bean based diets have the lowest and the soya based diets have the highest environmental impacts per kg pig. Both the pea and lupin based diets were concluded to have equal environmental impacts per kg pig.

Contents

List of Figures

List of Tables

Table of Abbreviations

ANF	Anti Nutritional Factor
CP	Crude Protein
DE	Digestible Energy
DNDC	Denitrification-Decomposition
EA	East Anglia
GAP	Good Agricultural Practice
GHG	Green House Gas
GWP	Global Warming Potential
IPCC	Intergovernmental Panel of Climate Change
LAI	Leaf Area Index
LCA	Life Cycle Assessment
ME	Metabolisable Energy
MJ	Mega Joules
NDF	Neutral Detergent Fibre
OA	Organic Agriculture
RL	Red Label
SAA	Synthetic Amino Acid
SALSA	Systems Analysis for Sustainable Agriculture
SCL	Silty Clay Loam
SOC	Soil Organic Carbon
UK	United Kingdom
Y	Yorkshire

Introduction

In order to remain competitive in the global market, comply with government policy that promotes sustainable pig farming and reduce environmental impacts, the British pig industry must seek viable and sustainable solutions to reduce environmental impacts when sourcing feed ingredients whilst maintaining a desirable level of output. Such sustainability may be enhanced by increased utilization of UK grown feed ingredients, as opposed to relying on imported feedstuffs. This has the potential to reduce energy demands and the environmental impacts of pig production. Whilst this can be relatively easily accomplished for the cereal component, the choice of home grown proteins is limited. In the UK's temperate climate, home grown protein sources are restricted to oilseeds (e.g. rape and sunflower seeds) and legumes (peas, beans and lupins).

The aim of the project is to quantify the environmental impacts of producing grower-finisher pigs (12 kg – 105 kg) using different diet scenarios. This will be carried out by developing a Life Cycle Assessment (LCA) model to include all processes involved in the production system. The diet scenarios included are; (1) conventional soya based diet, (2) home grown bean based diet, (3) home grown pea based diet and (4) home grown lupin based diet. Two UK sites are used in the LCA which have the highest proportion of pigs in the UK; East Anglia and Yorkshire. Two further scenarios will be used which consist of the same soil type in both East Anglia and Yorkshire and are modelled to determine the effects of climate only. The crops required for the pig diets will be theoretically grown at all four sites.

Three environmental impacts are assessed, these are: Global Warming Potential (GWP), eutrophication potential and acidification potential. The four diet and site scenarios will be compared for each environmental impact category to determine which scenarios fair better in environmental terms.

CHAPTER I

1 Critical Review of Existing Environmental Assessments of Pig Production Systems

1.1 Introduction

Environmental assessments of production systems are performed in all types of industries to determine their impacts on the global environment and to assess potential options to reduce the environmental impacts. The agriculture industry is actively seeking viable solutions to reduce its negative environmental impacts by investigating areas within production systems where the main environmental impacts occur. This is then used to determine potential mitigation options.

The pig sector is one of the agricultural industries aiming to lower its environmental impacts. Pig meat is in high demand and globally 1.3 billion pigs (Compassion in World Farming 2009) are slaughtered annually for human food consumption. Pig production systems (along with other livestock systems) rely heavily on natural resources, which includes land and water but also fossil fuels. Efforts to reduce the negative environmental impacts are now being made by producers whilst still aiming to maintain high level of outputs.

1.2 Environmental Assessment

Environmental assessment models are a means to quantify the environmental impacts associated with production systems. Several types of environmental assessments have been applied in different industry sectors. Each assessment method implements different approaches and levels of detail. Assessment methods include: Ecological Footprint, Nutrient Balance, Environmental Risk Mapping, Multi Agent System and Multi Linear Programming Approach. This review will summarize each of these approaches in turn and will discuss the advantages and disadvantages of each method before the Life Cycle Assessment (LCA) is introduced as a more comprehensive form of environmental analysis. The use of LCAs in different countries will be outlined, which will lead to the proposition that the limitations of LCAs in their current format reduce the validity of international, cross LCA comparisons. An alternative approach will then be suggested that draws on the strengths of these various models whilst minimizing the weaknesses associated with each.

1.2.1 Ecological Footprint

Ecological Footprint is a method used to indicate the human demand on the environment which was developed by Wackernagel and Rees (1997) to assess the impact an individual(s) has on the Earth's resources by quantifying the amount of nature they occupy in order to live. The assessment method is used to quantify the human populations demand for natural resources and the ability of the biosphere to regenerate the consumed resources (Wackernagel & Rees 1997; Herva *et al*. 2008; Niccolucci *et al*. 2008). Within the Ecological Footprint, five categories are included: consumed land, gardens, crop land, pasture land and productive forest. The potential sustainability of manufacturing the product is also considered. However, local, regional or global differences are not included, thus, the assumption is made that all land and water areas are the same. The method also only considers CO_2 and leaves other Greenhouse Gases (GHGs) which may have a profound effect on the environment outside the assessment. These are the identified weaknesses in the assessment, although there maybe scope to enhance the methodology to include other GHGs to give a more complete assessment of a system. Consequently, Van den Bergh and Verbruggen (1999) suggest that the potential environmental impacts of a production process are generally underestimated within the current method of Ecological Footprint.

1.2.2 Nutrient Balance

Another method of environmental analysis is Nutrient Balance (de Boer 2003). This is carried out to identify inefficiencies within a production system. At farm level, this approach assesses nutrient losses, erosion and leaching from a system. Nutrient

Balance focuses mainly on the assessment of N, P and K (macro - nutrients), therefore it can be useful when considering the environmental effects of crop production with regards to losses from applied fertilizers (nutrient loss = input – output). However, other inputs into the system are not considered, for example, fossil energy required for the manufacture of fertilizers. Nutrient Balance does not take into account site specific conditions. For instance, an assumption is included in the model for equal efficiencies of N, P and K applied to crops on different sites (Sheldrick & Lingard 2004). The success of Nutrient Balance varies, depending on the detail included in the assessment and the construction of the analysis for nutrient losses at farm level (de Boer 2003). It is not suitable, however, for assessment of all environmental burdens.

1.2.3 Environmental Risk Mapping

Another more holistic approach is Environmental Risk Mapping. This defines the environmental risks resulting from human pressure (for example farming practices) and from the vulnerability of the environment in a given region. However, it only assesses one impact category at a time, for example nitrate leaching or the transfer of phosphorus (Assimakopoulos *et al*. 2003; Payraudeau & van der Werf 2005). The assessment is constructed using several variables, and has the ability to include output data from simulation models. This assessment could be applied to determine the environmental impacts arising from a particular aspect of the farm system, for example manure management but it is not a useful tool to quantify several environmental impacts in the same assessment (Payraudeau & van der Werf 2005).

1.2.4 Multi-Agent System

The Multi-Agent System is an environmental assessment method that assesses the economic, social and environmental interactions of an agricultural system. The aim of this approach is to represent the behaviour of a defined group towards a limited resource and to calculate the use of resources within a system. It also allows different potential situations to be modelled. The parameters included can be controlled to determine the best possible scenarios for the management of the assessed product. Courdier *et al* (2002) and Payraudeau & van der Werf (2005) applied this method to analyze the management systems of manure and the overall impacts this caused to the sustainability of the environment. The aim of their study was to show different 'what if' scenarios for management of animal wastes. This approach is therefore advantageous when determining ways to reduce environmental impacts in a theoretical approach. However, the results are focused not on the environmental impacts *per se*, but rather directed at the social and economic impacts. Therefore, to make a thorough environmental assessment of a production system, Multi-Agent System may not be the most appropriate approach.

1.2.5 Multi Linear Programming

Multi Linear Programming is an approach that is applied to determine potential ways to minimize the environmental impacts of a system by optimizing production by considering its technical, economic and social aspects (Bouman *et al*. 1999; Payraudeau & van der Werf 2005). Linear optimization techniques are used to identify the management method(s) which maximizes profitability with minimal environmental emission (Payraudeau & van der Werf 2005). Indicators within the assessment can be adjusted to determine (theoretically) the best scenario within the system to reduce the environmental burdens. This approach is constructed in stages. Initially, the production system is described to include all inputs and outputs and all associated emissions. All environmental (also if required economic, agronomic and social) constraints are then incorporated which limit the management of the system. Finally, to determine the best possible scenario, linear optimization techniques are applied to determine the most appropriate management scenario which has the lowest environmental impact.

1.2.6 Life Cycle Assessment

Although the above approaches can be useful on a smaller scale, with the exception of Ecological Footprinting, to identify the environmental impacts within particular aspects of the pig industry, a more comprehensive environmental assessment is required to assess the environmental impacts of a complete production system. This identified approach is called Life Cycle Assessment (LCA). LCA includes all processes within the production system. It quantifies the GHG emissions and also other environmental impacts (such as eutrophication and acidification) associated with a product from cradle to grave (Brentrup 2004). LCA allows all processes within the life cycle, in this case pig production for food consumption, to be quantified to a specific functional unit. LCA is a more robust tool for assessing the environmental impacts as it has unlimited potential to include vast amounts of detail for all parts of the system. In many LCAs, the system is separated into sections and sub-models can be developed for each part of the system. The level of detail included does vary between LCAs, depending on what the assessor wants to achieve. Therefore each LCA is unique and awareness of this is important.

The International Organization of Standardization (ISO) has successfully introduced a protocol for the standardization of the structure of LCAs. Part of this standardization requires LCAs to be developed in stages, which includes (1) Goal and Scope, (2) Life Cycle Inventory (analyses), (3) Life Cycle Impact Assessment, and (4) Interpretation (Rebitzer *et al.* 2004). However, the precise methodologies used within each of the stages are not standardized. This can cause confusion and interpretative errors when comparisons are made between LCAs assessing similar systems.

1.3 A Review of LCAs for Pig Production Systems

LCAs aim to determine the environmental impacts of production systems by quantifying the environmental effects of all processes within the systems. The most common impact categories assessed are global warming potential (GWP), eutrophication and acidification potential which are expressed using a common functional unit. Different LCAs have been developed for similar production systems but since the level of detail may vary considerably, a direct comparison of results between LCAs is difficult. Variations of results occur mainly due to differences in

(1) methodologies, (2) system boundaries, (3) data included and (4) functional unit. Therefore, it is important to critically assess the construction of the LCA and the methodologies used before comparisons of results are made. Therefore, one of the aims of this review is to compare the methodologies and assumptions of five existing LCAs which have been developed for pig production systems.

The five LCAs referred to in this review assess the environmental impacts of pig production systems. These include firstly, a LCA developed for the Canadian pig industry which determined regional and historical differences, by comparing the long term trends from 1981 to 2001 (Verge *et al*. 2009). Secondly, a Danish LCA (Dalgaard *et al*. 2007) developed to assess pork production within Denmark. This included the additional environmental impacts occurring from transport to the Netherlands and to the United Kingdom. Thirdly, a UK Defra funded LCA (Williams *et al*. 2006), which was developed to quantify the environmental burdens and required resources for ten agricultural commodities, one of which was UK pig production. The fourth, a Swedish LCA, which assessed the production systems of grower-finisher pigs at farm level comparing three diet scenarios (Eriksson 2004). Then finally, the fifth LCA for French pig production systems, which compared three management scenarios, (1) Good Agricultural Practice (GAP) (intensive production), (2) Red Label (RL) (a quality assurance scheme) and (3) Organic Agriculture (OA) (Basset-Mens & van der Werf 2005). Each LCA was divided into five common categories to enable comparisons to be made. These categories were (1) pig growth and management, (2) slurry storage, (3) crop production, (4) soil and climate and (5) fossil fuel usage.

The Intergovernmental Panel of Climate Change (IPCC) is a global scientific body that communicates the current climate change situation to the world based on the most recent scientific research. IPCC also offer a consistent level of standardization within LCAs. This includes, firstly, the universal use of characterization factors to convert Greenhouse Gases (GHG) to CO_2 equivalents on a given time scale; 20, 100 and 500 years. And secondly, categorizing the methodologies used by the level of detail modelled from Tier I to Tier III. Tier I refers to the simplest method, for example basic assumptions and data are used in emission calculations. Tier II methodology is more complex than Tier I, assumptions used are country specific. On the other hand, Tier III uses the most detailed assumptions and modelling techniques and data used is country specific.

Table 1 IPCC equivalency of methodology used within different sections of the assessment for each reviewed LCA.

	Pig growth and management	**Slurry storage**	**Crop Production**	**Soil and climate**	**Fossil fuel usage**
Canada	Tier III	Tier II	Tier III	Tier I	Tier III
Denmark	Tier II (existing data used)	Tier II	Tier II (existing data used)	X (not clear as existing data used)	Tier II (existing data used)
UK	Tier III	Tier II	Tier II / III	Tier II / III	Tier II
Sweden	Tier II / III	Tier II / III	Tier III	Tier I	Tier II
France	Tier I (published statistic used)	Tier I	Tier I (not assumed grown on farm)	X (not clear as existing data used)	Tier I

It is important to be aware of the level of detail used to make a successful comparison between LCAs. Using the IPCC information, the methodologies used for each component of the system in the reviewed LCAs has been represented as IPCC Tier equivalents and are represented in Table 1. Table 2 highlights the differences in the level of detail used in the LCAs under consideration. For each LCA the data is sourced differently, this includes for example inventories or specific models. Therefore, based on the information published for each LCA, the IPCC Tier equivalent was determined. This demonstrates the difficulties involved in making comparisons between approaches by highlighting the similarities and differences of the five LCAs.

Table 2 Summary of the processes which were included in each LCA for pig production systems.

	Data adapted from existing livestock LCAs	Data sourced from inventories (all or part of the LCA)	Energy included for heating, lighting and ventilation	Energy included for crop production (management practices)	Energy included for production of machinery included	Fossil fuel use included	Transport included	Synthetic fertilizer included	Slurry applied as fertilizer	Slurry used in anaerobic digester
Canada	√	√	√	√	√	√	√ (farm transport)	√	√	X
Denmark	X	√	√	√	X	√	√ (slaughter house, slurry, feed and product destination)	?	√	√ (energy used to supply lighting)
UK	X	√	√	√	√	√	√ (soya transport from Brazil and Argentina)	√	√	X
Sweden	X	√	√	√	X	√	√ (soya transport from Brazil to Sweden)	√	√	X
France	X	√	√	√	√	√	√ (feed to farm, building materials)	√	√	X

√ = included
X = not included

1.3.1 Aims of Each LCA

Although the aims of each study varied, the end points are similar in all. This common end point is the determination of the environmental impacts of each pig production system. Each LCA assessed the GWP, eutrophication and acidification potential, with the exception of the Canadian LCA, which only assessed GWP.

The Canadian and French studies applied an historical approach. The Canadian LCA assessed the period from 1981 to 2001 and the French from 1996 to 2001. Although they both applied an historical assessment technique, the factors included within the frame work differed. This variation reflected differences in the purpose of the LCA. The aim of the Canadian study was to determine the regional differences within the pig industry to produce 1 kg of pig live weight, whereas the French assessed the different management systems using the functional units of 1 kg of pig live weight and per 1 hectare (Basset-Mens & van der Werf 2005). The aim of the Danish and the Swedish LCAs was to determine the effects of transport within the system. The Danish determined the effects of transporting the final product to three locations (1) Denmark, (2) the United Kingdom and (3) the Netherlands. By comparison, the Swedish study focused on the environmental costs associated with the transport of importing soya *per se*. This additional transport cost was compared with the use of two home grown protein sources: peas and rapeseed. The three diet scenarios included the use of (1) imported soya, (2) home grown peas and rapeseed cake and (3) Swedish produced peas and rapeseed cake with additional synthetic amino acids (Eriksson 2004).

The UK LCA differs from all other LCAs by not assessing the environmental impacts of differing management systems *per se*. Instead, it assessed the UK pig industry as a whole at commodity level. This was similar to the approach adopted in the Canadian LCA. Therefore, in the UK LCA the overall environmental impacts per 1 kg pig carcass weight included several management systems and finishing weights (see section 1.3.2). This LCA also included additional impact categories which were not included in the other LCAs. These were: abiotic resource use (the use of natural resources calculated on a common scale), primary energy use (diesel, electricity and gas quantified as MJ primary energy as coal, natural gas, oil and uranium required for nuclear electricity) and land use (for crop production). Additionally GWP per functional unit was assessed using the time scales of 20, 100 and 500 years. Within this review only the results for the 100 year time scale will be considered as this is the common time scale used in all reviewed LCAs.

1.3.2 System Boundaries of Each LCA

When comparing the LCAs, it is crucial to understand the system boundaries and what is included within the assessment. For each of the reviewed LCAs the boundaries are unique to the system under assessment. The system boundaries of each LCA begin and end at different points in the production cycle. These are shown in Table 3.

Table 3 Start and finish points used in the system boundaries.

	Start	**Finish**
Canada	Birth	85 kg (live weight)
Denmark	Sow and piglet	105 kg (live weight)
UK	Sow and Piglet	76, 87 and 109 kg
Sweden	29 kg	115 kg (live weight)
France:		
GAP	25.7 days	175 days
RL	28 days	190 days
OA	42 days	195 days

The Danish and UK LCAs use the same starting point. However, the exact weights and periods of time considered during this period are not given. Instead, this is given as the sow and piglet stage, allowing a wider view of the environmental impacts of pig production. The system boundaries of the Canadian LCA began at birth and included all process until the pig reaches slaughter weight. The assessment finished when the pig reached a live weight of 85 kg. The methodology applied was initially developed for LCAs in the Canadian beef and dairy sectors (Verge *et al*. 2007; Verge *et al*. 2008). The Danish LCA described all processes within the life cycle until the pig reached 105 kg. These included all environmental cost of the processes which occur in the slaughter house and the system boundary closed when the product was delivered to the Port of Harwich in the UK. For the UK the system boundaries ended with one of three finishing (dead) weights 76, 87 and 109 kg. Indoor and outdoor systems were modelled (the organic system was assumed to be completely outdoors), and all finishing units were assumed to be indoors. The Swedish LCA began with a young pig weighing 29 kg. All environmental costs associated with feed production were included and the LCA boundaries closed when the pig reached a slaughter weight of 115 kg (the transport of the pig to the slaughter house was also included). The starting and finishing weights used in the French LCA differed for each management scenario and the boundaries included the processes up to and including pig production on farm (Basset-Mens & van der Werf 2005).

1.3.3 Diets

When assessing the environmental impacts of pig production systems, the diet production constitutes the most to the total. This depends not only on the diet compositions, but also on the production of the feed ingredients. Therefore it is necessary to include the environmental impacts of all diets within the system. All of the LCAs used conventional soybean meal as a source of protein in at least one diet scenario. The Swedish LCA did however incorporate several diet scenarios, one of which used only home grown peas (supplemented with rapeseed).

1.3.4 Methodologies

IPCC methodologies and emission factors used in the Canadian LCA ranged from Tier I (equivalent) to Tier III (equivalent) (Table 1). The IPCC emission factors were adjusted to Canadian conditions, for example soil water availability was calculated using the ratio of precipitation to potential evapotranspiration during the growing season in Canada. It included modifications for the influence of land characteristics, soil properties and tillage processes. This was achieved using an established model that simulates a complete set of farm operations on a variety of theoretical farms (Janzen *et al*. 2003; Verge *et al*. 2009) and an additional leaching fraction is determined (Verge *et al*. 2009). IPCC Tier II methodologies were used for the calculations of CH_4 from manure by incorporating the volume of feed intake and the fraction which is digested. The manure storage system used in the LCA was slurry tanks. Enteric CH_4 was included, again using IPCC Tier II equivalent. 1.5 kg CH_4 per pig per year, which was indexed to each swine category using the ratio of the category weight and the average weight (Verge *et al*. 2007; Verge *et al*. 2008). These values were also used in the UK LCA (Williams *et al*. 2006). The production of inputs into the systems was also included in the LCA, for example the manufacture of synthetic fertilizers was calculated using the conversion factor 4.1 kg CO_2 equivalent^{100} per 1 kg of synthetic fertilizer, independent to type of fertilizers (Verge *et al*. 2009). The formula used in the Canadian beef LCA to calculate emissions from electrical energy required for housing was derived from Dyer and Desjardins (2003) which was adjusted to Canadian conditions to estimate electrical energy. This was equivalent to Tier II methodology as the size of the unit, the size of pig and the number of pigs in the unit were factored into the equation. The environmental effects of producing pigs ended when they reached slaughter weight and therefore no environmental costs were included for processes involved at the slaughter house.

In contrast to the Canadian LCA, the Danish LCA consistently used Tier II equivalent methodologies. The energy use required for feed production and housing are included. The electrical energy is assumed to supply heating, ventilation and lighting; data was sourced using nationally derived figures from the LCA Food Database (www.lcafood.dk).

Transportation of pigs to the slaughter house was included within the system boundaries and the assumption was made that this was 80km when the pig reached 105 kg (79.2 kg carcass weight) (Dalgaard *et al*. 2007). A mass balance approach was adopted in the Danish LCA to account for N_2O emissions from manure fertilizer by assuming all N that enters the pig as feed leaves as either N in pig meat or as N in manure (Dalgaard *et al*. 2007). IPCC (2006) emission factors (Tier 1 equivalent) were applied to calculate N_2O emissions from slurry (46g of N_2O per 100 kg live weight). This approach is simpler than that used in the Canadian LCA. Finally, the carcass is assumed to be transported by lorry from the slaughter house, which has been assumed to be 126km in Denmark and an additional 619km to the UK. The data for this was sourced from the Ecoinvent Centre (2004) (Dalgaard *et al*. 2007).

The UK LCA was developed to represent environmental effects of the UK pig industry as a whole, therefore sufficient data was required to represent the various management systems. Data was sourced from established UK inventories by Williams *et al* (2006). All energy requirements for farm processes, for example diesel for farm operations within the system were included and traced back to their primary source (Williams *et al*. 2006). Since this category is not included in any other of the reviewed LCAs, this UK LCA could be considered more representative of the total environmental impacts from the production system. Similarly to the Danish LCA, the production of synthetic fertilizers, the additional costs of buildings and machinery required on farm were included which were also calculated in the Canadian and French LCAs. The soil nutritional content was assumed to be in steady-state when crops were grown; hence the soil carbon (SOC) was assumed to be in equilibrium (Williams *et al*. 2006). This was achieved by running the model SUNDIAL (Smith *et al.* 1997) for an unspecified time until the soil reached a steady-state (IPCC Tier III methodology). SUNDIAL was also used to predict the effects of rainfall on leaching and denitrification (Smith *et al*. 1997; Williams *et al*. 2006). The burdens associated with the proportions of crops used in the diets were allocated based on their economic value to quantify the impacts from the part of the crop used in the diet (Williams *et al*. 2006). Economic allocation of crop burdens was also implemented in the Swedish and French LCAs; however, a mass allocation approach was applied in the Canadian assessment. The Danish LCA did not state the method used. Finally, the energy use required for grain processing was included (drying, cooling and storing) (Williams *et al*. 2006).

The Swedish LCA is composed of three empirical Systems Analysis for Sustainable Agriculture (SALSA) models, which describe the substance and energy flows for each part of the production system (Eriksson 2004). The empirical models used were, (1) SALSA Arable, (2) SALSA Soya and (3) SALSA Pig. SALSA Arable assessed Swedish production of wheat, barley, peas and rapeseed (Eriksson 2004) and includes field operations, air emissions from crops, indirect N_2O emissions, drying of grain, pressing of rapeseed oil, electricity and diesel production, synthetic fertilizer production and seed production (Eriksson 2004). The SALSA Soya sub-model was used to assess the environmental impacts of soya production. The model includes the production of soya in Brazil including all extraction and proportions of the crop used in animal feeds and the transport to Sweden, which is determined by economic allocation, i.e. oil and soybean meal 69% and 31% respectively. The model also calculated the fertilizing effects of pig slurry converting this to avoided use of the corresponding amounts of synthetic fertilizer application. Additionally, emissions from the production and use of synthetic N fertilizers are also included. The third sub-model, SALSA Pig, includes the processes which take place at the pig farm: energy use for operation of buildings, emissions originating from animals and excreta in the barn, emissions from manure storage, and application to fields. The latter includes both from the manure itself and emissions from tractors required to spread the slurry (Eriksson 2004; Eriksson & Nybrant 2004). IPCC (2001) Tier I conversion factors were used to convert the gases produced from the system to quantify them for each environmental impact category. The three SALSA models were integrated to determine the environmental effects of producing 1 kg pig live weight.

The French LCA was constructed using a detailed inventory from the French pig industry, using data from French pig farmers, published literature data and existing data inventories. The production scenarios allow for the comparison of the environmental effects of systems which are better; for pig welfare, the environment or most profitable for the farmer. These factors have not been considered in any of the other LCAs. The assumption was made that feed ingredients were produced in Bretagne for GAP and RL and sourced from local producers, it was also assumed that all manure produced by the pigs was applied as fertilizer to the crops (Basset-Mens & van der Werf 2005). For GAP and RL the transport distance of 100km was allocated to French produced crops from the field to the processing plant and then finally to the pig farm (Basset-Mens & van der Werf 2005) and for OA a further transport cost was allocated this distance which was assumed to be 150 km (Basset-Mens & van der Werf 2005), due to the wider range of crops included in the diet which resulted in a requirement for further local transport. For feed ingredients not produced in France, soya was sourced from Brazil, sunflower from Argentina, cane molasses from Pakistan and cassava from Thailand (Basset-Mens & van der Werf 2005). Furthermore, resource use and emissions associated with buildings (production and delivery of materials, construction) were taken from previously published data for France (Basset-Mens & van der Werf 2005) and NH_3 and N_2O emissions were calculated from both published literature data and estimated emission values using IPCC data (Tier I methodology) (Basset-Mens & van der Werf 2005). To allow for a more robust LCA, uncertainty analysis was carried out on the most relevant issues namely: (1) crop yields and feed to gain ratio from weaning to slaughter (2) field emissions and leachates (NH_3, N_2O and NO_3) and weaning to slaughter leachate of NH_3 and N_2O from buildings and (3) manure storage and composting. This type of analysis has not been performed in any of the other LCAs considered.

1.3.5 Results of the Reviewed LCAs

The final stage in any LCA is interpreting the results to conclude the environmental impacts of the functional unit. Table 4 represents the results for each LCA described in this review.

Table 4 The results of each reviewed LCA for all impact categories per 1 kg of pig.

	GWP (**kg CO_2 equivalent^{100}**)	**Eutrophication** (**kg PO_4 equivalent**)	**Acidification** (**kg SO_2 equivalent**)
Canada:			
1981	2.98	—	—
2001	2.31		
Denmark	3.60	0.23 (kg NO_3 equ.)	0.045
Delivered:			
UK	3.60	0.30 (kg NO_3 equ.)	0.064
Netherlands	3.60	0.22 (kg NO_3 equ.)	0.042
UK	6.40	0.10	0.340
Sweden:			
Soya	1.47	0.55 (kg O_2 equ.)	0.024
Pea	1.31	0.55 (kg O_2 equ.)	0.025
Rapeseed & SAA	1.38	0.45 (kg O_2 equ.)	0.019
France:			
GAP	2.30	0.02	0.040
RL	3.46	0.02	0.020
OA	3.97	0.02	0.040

1.3.5.1 GWP

GWP arises from three GHGs (1) CO_2, (2) N_2O and (3) CH_4, which are expressed as CO_2 equivalents. All studies used IPCC GHG equivalency factors, however not all studies used the same version. The Danish and the UK studies used the 2001 factors:

CO_2 1, N_2O 296 and CH_4 23. Whereas, the French and Danish LCAs used the 1996 equivalency factors: CO_2 1, N_2O 310 and CH_4 21.

There are clear variations between the results of each LCA, with values ranging from 1.31 to 6.40 kg CO_2 equivalent^{100} per kg pig. The UK predicted the highest GWP; 6.40 kg CO_2 equivalent^{100} (latest result in 2009 after adjustments to the initial 2006 LCA). The Swedish LCA resulted in the lowest GWPs when compared with all other countries. Thus, for the pea based diet the GWP was calculated as 1.31 kg CO_2 equivalent^{100} and out of all three diet scenarios in the Swedish LCA, the soya based diet resulted in the highest GWP 1.47 kg CO_2 equivalent^{100} (Eriksson *et al*. 2004). The average GWP in the Canadian LCA for all provinces in 1981 was 2.98 kg CO_2 equivalent^{100} compared with 2.31 kg CO_2 equivalent^{100} in 2001. The lower GWP in 2001 was primarily due to changes in diets as the modern diets were composed of more digestible feed ingredients and therefore utilized by the pig more efficiently resulting in less N excreted per unit N intake (Verge *et al*. 2009). In addition to improved diets, pigs had higher birth rates and a lower marketing age in 2001 compared with pigs used in systems in 1981 and the proportion of the total population that were weaner pigs in 1981 dropped by 3% in 2001 (Verge *et al*. 2009). There was also a reduction in the amount of N fertilizer applied to crops in the 2001 scenario, which reduced the amount of N_2O emitted from soils, however N fertilizers still attributed most to the N_2O losses within the system (Verge *et al*. 2009). The GWP results for France ranged between 2.30 kg CO_2 equivalent^{100} for the GAP system and 3.97 kg CO_2 equivalent^{100} for OA, clearly showing that the management systems does affect the overall environmental impacts. In contrast, the GWP for pig production in Denmark was the same for all final destinations of Danish pork, i.e. 3.6 kg CO_2 equivalent^{100}. This identifies that transport does not impact considerably on the total GWP. The individual GHG contributions to total GWP were 44% for N_2O emissions, 32% for CH_4 emissions and 20% for CO_2 emissions.

It was concluded in the UK LCA that the main contributor to the environmental impacts occurred from feed production, more specifically from fertilizer application, which resulted in the highest amounts of N_2O emissions (Williams *et al*. 2006) and is thus a major contributor to the total GWP. This was also concluded in the Canadian, Swedish, French and Danish LCAs. More specifically, in the Swedish LCA, feed production was concluded responsible for 60 - 66% of the total GWP whilst 34 - 40% was attributed to the pig sub-system. In Denmark, the diet contributed more than 2.4 kg CO_2 equivalent^{100} of the total GWP (66.6%) and is therefore also considered the most significant contributor compared to any other areas of the production system, transport alone contributed only 1% to the total GWP and this also included the transport of soya from Brazil. In the French LCA crop production contributed to 54% of the total GWP in the OA scenario, increasing to 73% in the GAP scenario (Basset-Mens & van der Werf 2005).

With regards to individual crops in the Swedish LCA, peas had the lowest total energy requirement of all crops due to their N fixing ability (modelled as an avoided use of fertilizers), therefore concluded with the lowest GWP. Rapeseed meal had the lowest contribution to GWP; this is because rapeseed meal is a by-product from the rapeseed crop which is primarily grown for oil. Therefore, to calculate the environmental impacts from rapeseed meal, economic allocation was applied. Rapeseed meal is less valuable than rapeseed oil, therefore 30% of the total GWP of producing the crop was allocated to the rapeseed meal (Eriksson *et al*. 2004).

From the conclusions of the GWP uncertainty analyses in the French LCA, the biggest uncertainty occurred from field emissions relating to variations in yields and energy requirements from the crop (Basset-Mens & van der Werf 2005).

All of the studies, with the exception of the French LCA, concluded that transport contribute to ~ 1% of total GWP. However the French LCA concluded this to be 15 - 27% of total GWP, this therefore indicating how variations in methodologies can impact on the end result.

1.3.5.2 *Eutrophication*

It is important to note the difference in units used to measure eutrophication potential between the LCAs, this must be taken into account when comparisons are made between results. The Swedish LCA calculated the eutrophication potential in kg O_2 equivalent, which is not used in any other of the reviewed LCAs. The eutrophication results for the soya and pea based diets were the same, 0.55 kg O_2 equivalent. However the rapeseed and SAA based diet scenario resulted in a 20% lower eutrophication potential at 0.45 kg O_2 equivalent.

The Danish LCA used NO_3 equivalency and differences occurred for the eutrophication potential results to the various final destinations. Delivery of pork to the UK attributed to a higher eutrophication potential, 0.301 kg NO_3 equivalent when compared with pork production for Denmark, 0.232 kg NO_3 equivalent (an average decrease of 0.07 kg NO_3 equivalent or 23%) which is caused by the additional fossil fuel required for transportation. The delivery of pork to the Netherlands faired better in terms of eutrophication potential, resulting in 0.219 kg NO_3 equivalent due to the method of transport. Grain production contributed the most to eutrophication, 0.122 kg NO_3 equivalent. The contributions of gases to eutrophication are, NO_3, NH_3, NO_x and P leaching; 62%, 32%, 4% and 2% respectively.

Crop and feed production was also the main contributor to eutrophication potential in the OA and RL scenarios, 64% and 71% respectively (Basset-Mens & van der Werf 2005). An uncertainty analysis was performed within the LCA and for eutrophication, the biggest uncertainty occurred from field emissions relating to variations in yields and energy requirements from the crop (Basset-Mens & van der Werf 2005).

1.3.5.3 *Acidification*

The acidification potentials for each LCA ranged from 0.019 kg SO_2 equivalent to 0.064 kg SO_2 equivalent. In the Swedish study there is a variation of 25 % for the acidification potentials between diet scenarios, 0.024 kg SO_2 equivalent for the soya based diet, 0.025 kg SO_2 equivalent for the pea based diet and 0.019 kg SO_2 equivalent for the rapeseed and SAA based diet (Eriksson *et al*. 2004). For all impact categories, soybean meal was the highest contributor to acidification potential and in the soya based diet 75% of the total was due to the long distance transport of soybean meal (Eriksson *et al*. 2004). The pig sub-system NH_3 emissions contributed 78 – 88 % of the acidifying potential from manure. Sensitivity analysis was performed in the Swedish LCA on the impact of variation in feed conversion ratio through reducing the metabolisable energy (ME) of the diet by 10%, i.e. from 35 MJ to 31.5 MJ per kg live weight gain. This effectively reduced N excretion by 15% and acidification by 20 % whilst still assuming the same N retention. This would however vary with genotype but this indicates that pigs with a lower feed conversion ratio may have a positive effect on the environmental impacts of the system.

The acidification potentials calculated in the Danish LCA ranged from 0.064 kg SO_2 equivalent for the UK scenario and 0.042 kg SO_2 equivalent when pork is delivered to the Netherlands, this is an average increase of 0.02 kg SO_2 equivalent and NH_3 accounts for 84% of the total. In the French LCA, the highest acidification potentials occurred in the GAP and OA management systems, 0.04 kg SO_2 equivalent and RL concluded the lowest, 0.02 kg SO_2 equivalent. Crop production contributed between 24% and 34 % of the total acidification potential (Basset-Mens & van der Werf 2005). The outcome of the uncertainty analysis relating to acidification potential did not vary greatly between production systems and the main contributors from the system were due to the emissions from buildings and manure storage.

The results of the five reviewed LCAs clearly differ. However, variations in the methodologies used and differences in production systems for each country must be considered. Therefore, it is difficult to directly compare the environmental impact of producing pigs in one country with another.

1.3.6 Discussion

This review has identified the variations which occur between existing LCAs for pig production systems, and the reasons why these variations occur. All of the LCA frameworks follow the ISO 14040 standardization rules, therefore the structures of the LCAs are similar. The benefit of this standard framework allows the models to be divided into sections, for example, exploring at the inventory analysis stage so identification of where the input data has been sourced. However, this review also clearly shows the differing levels of detail included in each LCA, for example the detailed LCA for Swedish pig production implements specific sub-models for different parts of the system, which is then compared with the French LCA which uses literature data. The system boundaries are also important to consider before direct comparisons can be made. All LCAs do have limits with what can be assessed which could potentially be extended further. For example, this could include the breeding phase: boar(s) sow(s) and piglets required for maintaining the pig herd.

Although in each of the LCAs described the functional unit is one kg pig produced, variation in results may arise from whether this is live weight, carcass weight or product weight. Additionally, the initial starting point of the LCA can be very different (Table 3), which was highlighted in the UK and Danish LCA. Consequently, both incorporate the sow and piglet stage, allowing a broader and more detailed assessment of the complete life cycle assessed and not just the production of a pig from a given starting weight to slaughter weight.

In addition to the system boundaries varying between LCAs, the level of detail which is assessed within the boundaries is important. In the LCAs for pig production systems, the production of feed is the highest contributing component of the system. However, each LCA calculates the contributions of feed production differently. The UK and Swedish LCAs use simulation models to predict GHG emissions from crop production. In contrast, the Danish and French LCAs have used existing literature data, which does not include site specific data for the developed LCA. This could essentially result in wide variations within the results based on the assumptions that have been used. Differences between LCA results for similar parts of the system are also evident, for example it was concluded by the Danish, Swedish and the UK LCAs that transport contributes ~1% of the total GWP. However by contrast, the French LCA calculated a significantly higher percentage of 15 – 27 %. Again this is dependent on the assumptions applied and is also farm specific. Thus, the variations between the LCAs described makes it difficult to directly compare LCA results with each other. Although breaking the models down, as done in this review, allowed for an improved comparison of the used methodologies, it still does not allow effective comparison of the outcomes of the different LCAs.

1.3.7 Conclusion

To conclude, it is evident that variable results occur between LCAs which have been developed to assess similar production systems. Several reasons for these variable results have been identified, including differences in the boundaries applied to the LCA, the methodologies used in the LCA, the level of detail and the source of the data that was included and the functional unit used. The common agreement from the reviewed LCAs, is that feed production is the main contributor to the environmental burdens. Each LCA is unique to the system it is assessing and should be analyzed individually. For that reason, direct comparison of the environmental impacts between LCAs is not a valid assessment due to the extensive variations between methodologies. A more appropriate assessment method would be to therefore identify differences between the system boundaries and methodologies.

After reviewing the existing LCAs for pig production systems, it was proposed to develop a UK LCA to asses pig production systems at farm level and model the environmental impacts of several diet scenarios. To compare the environmental impacts of producing a completely UK home grown diet with the normal soya based diet. The aim is to develop a detailed LCA and include complex modelling of the major components of the systems; crop and animal production. The methodologies used in the development of this LCA will be described in Chapter II.

CHAPTER II

2 Description of the LCA Model

The LCA tool that will be described was developed to analyze the environmental impacts of the use of different protein sources in a typical UK pig production system. Environmental impacts are calculated and compared for several diet scenarios, i.e. a conventional (non-organic) soya based diet and UK home grown protein based diets using peas, beans or lupins. A holistic approach is used to incorporate all processes within the system boundaries from crop production and rearing of the pig to slaughter weight. The LCA is developed as a defined system and boundaries of the system are set accordingly. The LCA begins at two weeks post weaning, when the pigs reach 12 kg and the boundary is closed when the pig reaches a slaughter weight of 105 kg. The transport of the pig to the slaughter house and the slaughter process are not included within the boundaries.

The LCA will be compiled using simulation models and collected activity data will not be included for feed production. From the simulated models the environmental impacts associated with pig production will be predicted.

The LCA was developed using two dynamic and deterministic models that are integrated with the aid of additional sub-models describing all processes within the system boundaries. This allows the GWP, eutrophication and acidification potentials associated with the production of 1 kg of pig live weight to be predicted. The two main models used were (1) DeNitrificationDeComposition (DNDC) (li 2007a) and (2) Animal Growth Model (Emmans 1997 and Wellock *et al*. 2004). These models were selected because of their abilities to model the two major components of the pig production system, i.e. crop growth and pig growth. DNDC simulates crop growth within a rotation at a specified location and input parameters can be adjusted to represent each site. The crops which were simulated by DNDC in this study include all crops that were required to formulate the diets. The Animal Growth Model developed at SAC by Emmans (1997) and Wellock *et al* (2003) is used and parameters adjusted to represent current commercial pig breeds. All management processes involved within the system are included in additional sub-models; the models are subsequently integrated to calculate the potential environmental impacts of the pig production systems. The structure of the different models and the manner in which these are integrated is given in Figure 1.

2.1 Inventory

Table 5 Conversion factors used in the LCA

	Conversion Factor
IPCC GHG conversion factors:	
CO_2	* 1
CH_4	* 25
N_2O	* 298
1 MJ electricity	0.261 kg CO_2 equivalent[100]
1 litre diesel	2.6 kg CO_2 equivalent[100]
	0.000032 kg NO_x equivalent
	0.0000013 kg SO_x equivalent
1 kg N fertilizer	6.8 kg CO_2 equivalent[100]
	0.0005 kg PO_4 equivalent
	0.0047 kg SO_4 equivalent
1 kg P fertilizer	1.2 kg CO_2 equivalent[100]
	0.000074 kg PO_4 equivalent
	0.008 kg SO_4 equivalent
1 kg K fertilizer	5.7 kg CO_2 equivalent[100]
	0.00003 kg PO_4 equivalent
	0.0047 kg SO_4 equivalent
Herbicide and fungicide:	
1 dose per ha	8.0 kg CO_2 equivalent[100]
	0.015 kg PO_4 equivalent
	0.096 kg SO_4 equivalent
Transport:	
1 km road	0.000168 kg CO_2 equivalent[100]
1 km sea	0.0000106 kg CO_2 equivalent[100]
Energy required for grain processing:	
Cereals	0.26 MJ/kg
Rapeseed	0.31 MJ/kg
Soya	0.47 MJ/kg
Electricity required for housing per pig:	53 MJ
	0.261 kg CO_2 equivalent[100]
Production of feed additives:	
1 kg SAA	3.6 kg CO_2 equivalent[100]
	0.041 kg SO_4 equivalent
1 kg vitamins and minerals	0.4 kg CO_2 equivalent[100]
DNDC out puts C and N converted to GHGs:	
C to CO_2	* 3.67
C to CH_4	* 1.33
N to N_2O	* 1.57
Conversion from grain yield/ha to C yield/ha	* 0.45
Stored slurry:	
1 m^3	0.00179 kg NH_3

Table 6 Inventory analysis of the modelled sites

	East Anglia	Yorkshire	Brazil
Selection of site location			
Longitude:	51.8^o north	53.8 ^{o} north	10^o south
Soil type:	Clay loam and silty clay loam	Sandy clay loam and silty clay loam	Sand
Crop rotation:	Spring legume - spring barley - winter barley - winter rapeseed - winter wheat	Spring legume - spring barley - winter barley - winter rapeseed - winter wheat	Corn – soya

Table 7 Inventory analysis of the fertilizer scenarios

	Synthetic fertilizer scenario	Slurry fertilizer scenario
Fertilizers applied	NH_4NO_3 P_2O_5 K_2O	40 % NH_4NO_3 P_2O_5 K_2O 60 % Slurry
Slurry composition:		0.0042 kg total N 0.0017 kg P 0.0025 kg K
Stored slurry		
Bulk density:	-	1.03 kg/m^3

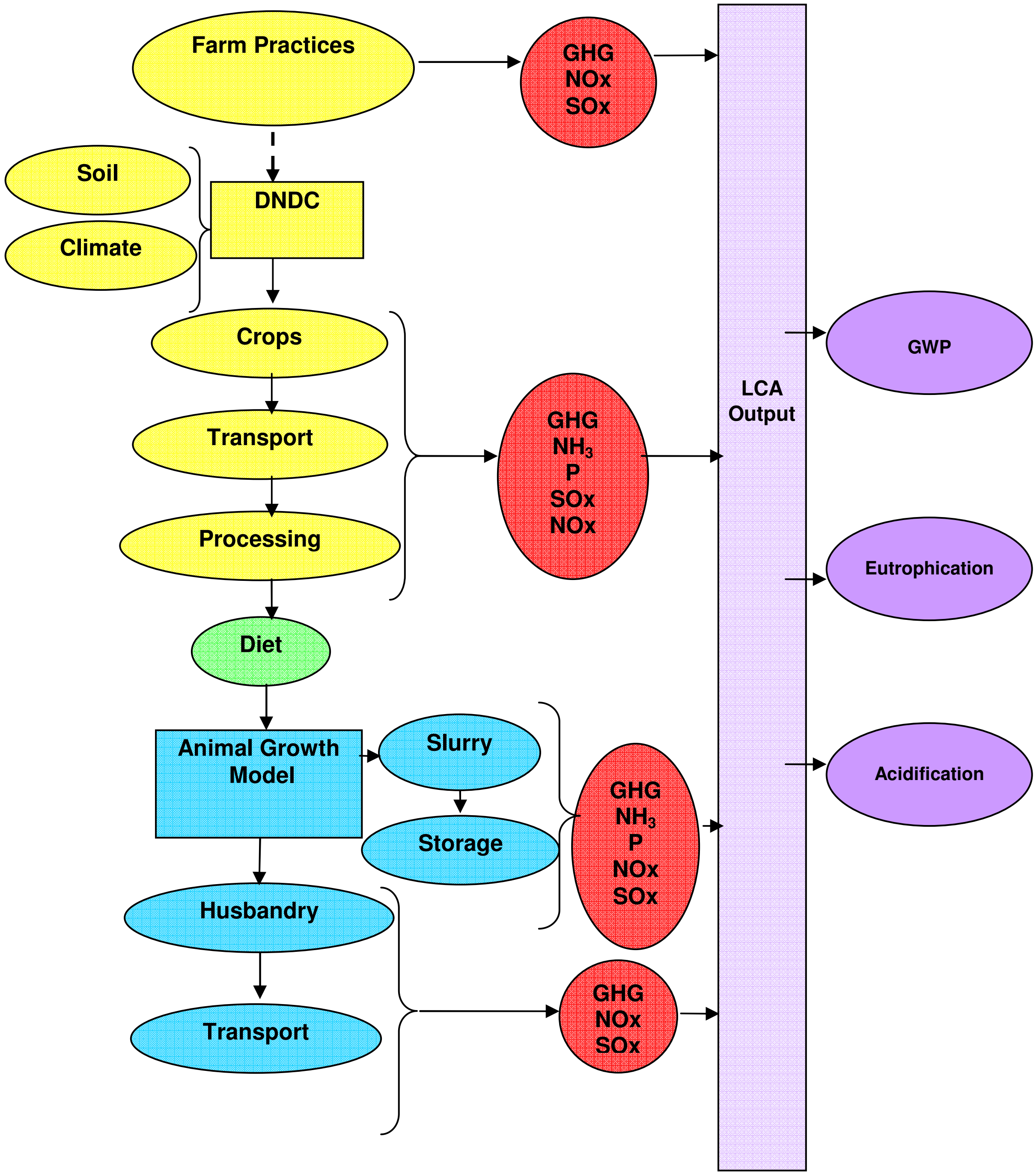

Figure 1 The LCA structure is built using the Denitrification DeComposition (DNDC) and Animal Growth Model (squares) with inputs and outputs represented by ovals. Yellow refers to the crop model and its immediate input and output variables, blue refers to the pig model and the diets (green oval) is related to both crop production and animal

growth. The red ovals are represented as outputs to determine the GWP, eutrophication and acidification potential.

2.2 Denitrification-Decomposition (DNDC) Model

The DNDC model is a process orientated computer simulation model of carbon (C) and nitrogen (N) biogeochemistry in agricultural ecosystems (li 2007a). The model consists of two components; the first consists of soil, climate, crop growth and decomposition sub-models, which predicts soil temperature, moisture, pH, redox potential and substrate concentration profiles driven by ecological drivers (e.g, climate, soil, vegetation and anthropogenic activity). The second component consists of nitrification, denitrification and fermentation sub-models, which predicts NO, N_2O, N_2, CH_4 and NH_3 fluxes based on the modelled soil environmental factors (Figure 2).

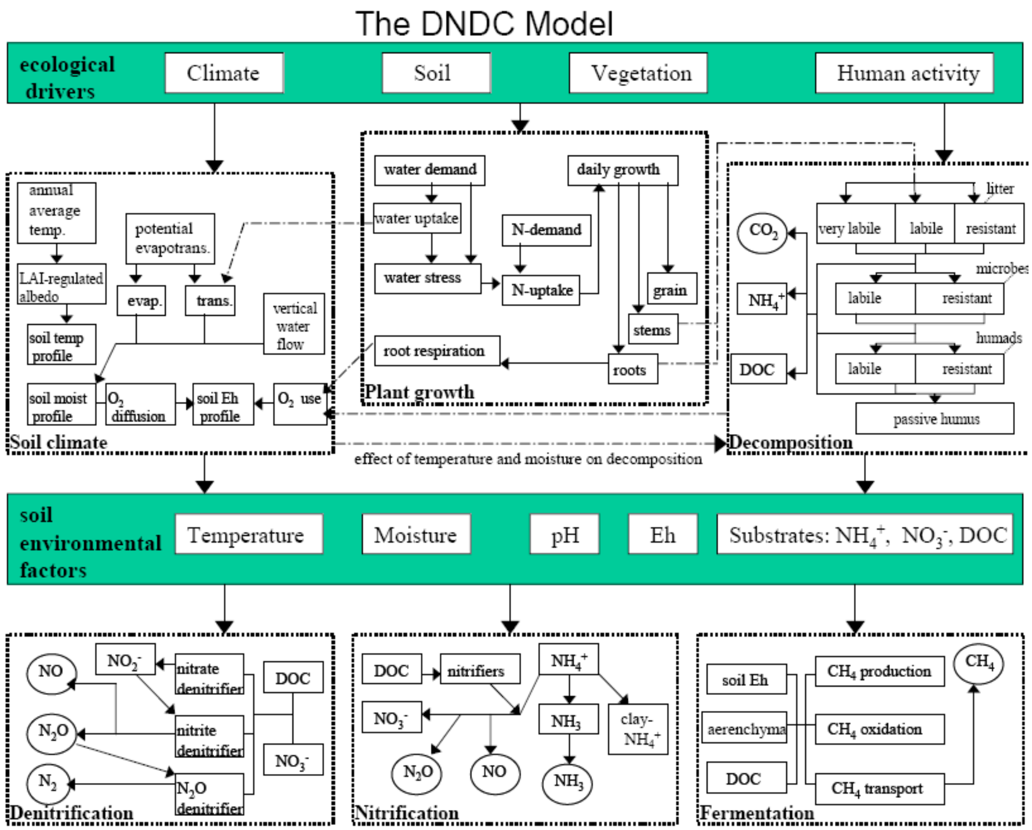

Figure 2 Detailed structure of DNDC, the green boxes represents the ecological drivers. Within the diagram, in separate squares are the sub-models which are required by DNDC to run effectively (Li, 2007b).

The two UK sites modelled in the LCA are East Anglia and Yorkshire as they have the highest pig populations in the UK (Defra 2006). The exact locations of the theoretical farms were selected from Defra statistics (Defra 2006), based on the highest population of pigs in that area and defined by their longitude and latitude. The specific locations are then used to source additional input data required for accurate simulation of crop growth within DNDC. A Brazilian soya farm is modelled to provide data on the environmental impacts of soya production. The required data for the theoretical farm along with the longitude and latitude was provided from personal communication with Prof. Da Silva (2009).

2.2.1 Inputs

2.2.1.1 Climate and Atmospheric Background

Daily weather data was sourced for East Anglia and Yorkshire from the British Atmospheric Data Centre website (BADC 2008) by selecting weather stations in the selected areas that were able to provide ten years of weather data for the period 1998 – 2007. In the case of Brazil daily weather data for 2007 was sourced from the National Climatic Data Centre (www.ncdc.noaa.gov). Ten years of daily weather could not be sourced due to the restricted availability of the data, this is identified as a limitation in the LCA. The following latitudes: 51.8^o north (East Anglia), 53.8 ^{o} north (Yorkshire) and 10^o south (Brazil) were used to specify the location of the sites in DNDC and to obtain exact weather data for that site. The weather data required for DNDC simulations include the maximum temperature (^{o} Celsius), minimum temperature (^{o} Celsius) and daily rainfall (cm) (see Figure A1 for an example of a weather file). DNDC requires information on the atmospheric concentrations values of ammonia (NH_3) and carbon dioxide (CO_2) which are defined in Table 5.

Table 8 Background atmospheric default data parameter used in DNDC.

Parameter	**Value**
N concentration in rainfall (mg N/L or ppm)	2.4
Atmospheric background NH_3 concentration (ug N/m^3)	0.06
Atmospheric background CO_2 concentration (ppm)	350

Li (2007)

2.2.1.2 Soil

Soil profiles are set up for each site and specific data for each soil type is entered allowing simulation of the growth of the crops. Data was sourced through the National Soil Research Institute soil maps (NSRI 2007) and soil types matched as closely as possible to the area (see Table 9). The selected soils were clay loam in East Anglia and sandy clay loam in Yorkshire. In addition, a soil type that occurs in both regions (silty clay loam) was modelled for each site to be able to investigate the effects of climate *per se* on the environmental impact of pig production. From advice by Prof Da Silva a sandy soil was selected for the simulation of soya and maize corn growth in Brazil.

Table 9 Parameters used in DNDC to define the soil profile at each site.

	East Anglia	**Yorkshire**	**Silty Clay Loam; East Anglia and Yorkshire**	**Brazil**
Soil texture	Clay loam	Sandy Clay Loam	Silty Clay Loam	Sand
Clay Fraction (0-1)	0.41	0.27	0.34	0.03
Bulk Density (g cm-^{3})	1.12	1.44	1.42	1.52
Field Capacity (water filled pore space (wfps), 0-1)	0.57	0.52	0.55	0.15
Hydroconductivity (m/hr)	0.0088	0.0226	0.015	0.6336
Soil pH	6.5	6.5	6.5	6.0
Wilting Point (wfps, 0-1)	0.27	0.24	0.26	0.1
Porosity (0-1)	0.476	0.421	0.477	0.395

(NSRI 2007)

The duration of the crop rotation is selected prior to the specific crop data being entered. For the UK, five year rotations are used (spring legume, spring barley, winter barley, winter rapeseed and winter wheat), repeated four times and for Brazil a two year rotation (corn, soya) which was repeated ten times. Hence, all scenarios include a twenty year simulation. The final five years of the UK rotations and the final two years of the Brazil rotations are used in the assessment. They are simulated for this length of time to allow stabilization of soil pools. DNDC requires specific site information for soil composition and the accuracy of the input data is essential for realistic representation of the modelled sites. Table 10 shows the initial input data with regards to soil composition.

Table 10 Soil organic carbon (SOC), initial NO_3 and NH_3 concentrations, microbial activity and slope parameters for the UK and Brazil sites.

	East Anglia	Yorkshire	Silty Clay Loam	Brazil
SOC at surface soil (0-5cm) (kg C/kg)	0.0312	0.0312	0.0312	0.03
Initial NO_3(-) concentration at surface soil (mg N/kg)	35	16.5	25.75	9
Initial NH_4 (+)concentration at surface soil (mg N/kg)	5.65	4.7	5.12	0.9
Microbial acivity index (0-1)	1	1	1	1
Slope	0	0	0	0

(Mahmood *et al*. 1998; Abbasi & Adams 2000; li 2007a; Da Silva 2009)

2.2.1.3 Farm Management

In each year, all crop data and timing of farm management practices are defined. This includes the crop type (selected from a default list of 49 crops) and detailed input data associated with the crop, N and water demands are entered and the values used in this LCA are given in Table 11. The planting and harvest dates of each crop for each rotation are shown in Table 12. The dates of the tillage practices required for each crop are shown in Table 13. Then finally, the application date, quantity and type of fertilizer applied, this includes both synthetic and slurry fertilizer scenarios (Table 14).

Table 11 The input parameters used in DNDC to define each crop. The light grey shaded parameter values represent values that are automatically calculated by DNDC, which is either based on the other parameters entered (including maximum biomass and biomass fraction) or by the selection of crop: total N demand. The fully shaded boxes represent the DNDC default parameters.

	Spring Barley	Winter Barley	Rapeseed	Wheat	Pea	Bean	Lupin	Corn	Soya
Max. biomass, kg C/ha									
Grain	3000	2300	2500	3800	2500	2500	2500	3000	2000
Leaf and Stem	4700	1880	7500	6741	3194	3194	3194	2412	850.1
Root	2300	800	869.5	1716	1250	1250	1250	590.7	377.8
Biomass fraction									
Grain	0.5	0.5	0.33	0.47	0.34	0.34	0.34	0.39	0.35
Leaf and Stem	0.4	0.4	0.55	0.38	0.48	0.48	0.48	0.49	0.45
Root	0.1	0.1	0.12	0.15	0.18	0.18	0.18	0.12	0.2
Biomass C:N									
Grain	40	40	20	25	18	18	18	35	10
Leaf and Stem	80	80	45	50	40	40	40	50	45
Root	80	80	52	50	35	35	35	50	24
N demand, kg N/ha	1125.0	150.0	235.1	237.7	264.9	264.9	264.9	114.9	100.8
Thermal, ^{o}days	1800	2500	2300	2000	1900	1900	1900	2550	1500
Water demand, g water/g DM	150	150	150	200	300	300	300	250	541
N fixation index	1	1	1	1	3	4	5	1	10
LAI adjustments factor (>0)	3	3	4	3	3	3	3	5	3

Table 12 Plant and harvest dates for each crop in the rotations for East Anglia, Yorkshire and Brazil.

East Anglia and Yorkshire	**Plant**	**Harvest**
Spring Peas/Beans/Lupins	1st March (year 1)	1st September (year 1)
Spring Barley	15th February (year 2)	5th August (year 2)
Winter Barley	1st October (year 2)	9th August (year 3)
Winter Rapeseed	20th September (year 3)	10th July (year 4)
Winter Wheat	2nd October (year 4)	20th August (year 5)
Brazil		
Corn (maize)	5th May (year 1)	4th October (year 1)
Soya	5th May (year 2)	27th October (year 2)

The UK rotation was selected from advice from SAC agronomists (SAC Farm management Handbook, 2009) and the Brazilian corn soya rotation was selected using advise from Prof da Silva (2009).

Table 13 Tilling methods for each crop at each site.

	Tillage	**Date**
East Anglia and Yorkshire		
Year 1	Plough with disc, 10cm	1st February
Year 2	Plough with disc, 10cm	15th January
	Plough with disc, 10cm	30th August
Year 3	Plough with disc, 10cm	12th August
Year 4	Plough with disc, 10cm	20th August
Year 5	-	-
Brazil		
Year 1	Plough with disc, 10cm	1st April
Year 2	Plough with disc, 10cm	1st April

Two distinct scenarios are used within the LCA regarding fertilizer application. The first is the use of synthetic fertilizers only and the second includes application of pig slurry. For each inorganic fertilizer application the method of application and the amount and type of fertilizer applied are required. Defra's RB209 fertilizer recommendation guide (Defra 2008) has been used to calculate the amount of N, P, K, fertilizer required by each crop (Table 14) in the rotation at each site. It is assumed that the N fertilizer type was ammonium nitrate (NH_4NO_3) P fertilizer is P_2O_5 and K_2O as K fertilizer (Defra 2008)

Table 14 Dates and quantity of synthetic fertilizer applications in East Anglia and Yorkshire. All fertilizer was applied at the surface.

	Date	N (kg/ha)	P (kg/ha)	K (kg/ha)
Spring Peas/Beans/Lupins	-	-	60	65
Spring Barley	20th April (year 2)	40	77.5	80
	1st June (year 2)	40		
Winter Barley	1st March (year 3)	40	77.5	80
	20th April (year 3)	110		
Winter Rapeseed	20th September (year 3)	30	75	65
	30th February (year 4)	80		
	1st April (year 4)	80		
Winter Wheat	10th May (year 5)	100	85	70

(Defra 2008)

The maximum grain biomass is specific for each crop and is a maximum potential value per hectare. This was calculated by taking the average yields/ha for each crop from the SAC Farm Management Handbook (2009). The grain yield was converted to C yield/ha by multiplying by 0.45, based on molecular weights of C and carbohydrates (Thornley 1998). It was assumed that the water content was 16% and therefore this value was then multiplied by 0.84 to convert to DM. DNDC requires the maximum potential yield of each crop which represents the yield of a non-stressed plant, therefore it is not limited by N, water or temperature. The yields given in the SAC Farm Management Handbook (2009) are approximate relay yields and therefore the plants will have suffered some stress. For this reason, when inputs were made into DNDC to achieve a value for maximum potential yield 1000 kg/C/ha was added and the values for leaf + stem and root are dependent on the maximum biomass for the grain value and are automatically calculated by DNDC.

The biomass fraction for grain and leaf + stem for each crop are calculated using data from Lecoeur & Sinclair (2001) and Kemanian *et al* (2007). The C:N for each crop is calculated individually which is related to the protein content of the crop which was sourced using feeding tables from Premier Nutrition Products (2005). Once the previously described input parameters have been entered, DNDC calculates the crop N demand. The N fixation index is entered for each crop; this is on a scale of 1 - 10 and is checked against the N fixing ability of each crop. For the legume crops, this is increased depending on the N fixing ability of the crop using data from Haynes *et al* (1993), Guafa *et al* (1993) and Unkovich & Pate (2000). This index equals 1 for all crops which do not fix N (Li 2007a). The Leaf Area Index (LAI) for each crop was defined on the basis of data from Darnmer *et al* (2008). The DNDC default value for vascularity of crops is 0 for all crops, as this parameter is only required for wetland crops.

In the second scenario, it is assumed that slurry is applied as fertilizer to feed crops. Slurry production is determined by the N excreted from pigs and is thus predicted by the Animal Growth Model. Each kg of pig slurry contains on average 0.0042 kg total N, 0.0017 kg P and 0.0025 kg K (Kyriazakis 2006). Using the assumption that 2.5 pigs are finished per year per pig farm, the slurry production was then calculated using these assumptions in linear equations. The average C:N ratio in pig slurry is 6 (Vallejo *et al*. 2006; Reijs *et al*. 2007) and the C concentration in slurry is therefore, estimated as:

$$\mathbf{6 * 0.0042 = 0.0252\ kg\ C/kg\ slurry}$$

which was used as an input parameter into DNDC. The slurry composition was then used to calculate the amount of slurry required to supply the N requirements for each crop grown in the rotation in replacement of synthetic fertilizers. Recommendations for slurry applications were taken from previous research at SAC. According to these recommendations, slurry applications were limited to 60% of N requirements as the plant only has access to the available N in the short term, with the remainder being met by synthetic fertilizers (see Table 15 and Table 16).

This was achieved by, firstly calculating the N required per hectare for each crop, followed by calculating 60% of this to determine the amount of available N to be supplied by slurry. The N content in the slurry is not completely available to the crops, and so the available N content of slurry is used to calculate total slurry requirements:

Total N slurry required (kg/ha) = N Req * Tot N/Avail N

After the slurry requirement per hectare is determined, the P and K application within slurry is calculated as

P supplied by slurry (kg/ha) = Slurry required to meet N requirement * 0.0017

K supplied by slurry (kg/ha) = Slurry required to meet N requirement * 0.0025

As only 60% of N requirement can be applied as slurry, the remaining 40% is applied as synthetic fertilizer. If the P and K requirements of crops are not met by the application of slurry, the amount of synthetic fertilizer to supply additional P and K is calculated as the difference between requirements and amounts supplied by slurry. The resulting slurry and synthetic fertilizer applications per crop and the relevant dates are given in Table 15 and Table 16.

Table 15 Amount of slurry applied to each crop per hectare (60% of total N requirement).

		Slurry	
	Date	Kg C ha^{-1}	Kg total N ha^{-1}
Spring Peas/Beans/Lupins	-	-	-
Spring Barley	20th April (year 2)	275	46
	1st June (year 2)	275	46
Winter Barley	1st March (year 3)	273	46
	20th April (year 3)	758	126
Winter Rapeseed	20th September (year 3)	195	33
	30th February (year 4)	547	92
	1st April (year 4)	547	92
Winter Wheat	10th May (year 5)	687	115

Table 16 The synthetic fertilizer requirements which is applied to each crop at the soil surface on a per hectare basis in the slurry fertilizer scenario (40% of total requirement)

East Anglia and Yorkshire	Date	N Kg ha^{-1}	P Kg ha^{-1}	K Kg ha^{-1}
Spring Peas/Beans/Lupins	30th May (year 1)	-	60	65
Spring Barley	20th April (year 2)	16	20.2	0
	1st June (year 2)	16	20.2	
Winter Barley	1st March (year 3)	16	4	0
	20th April (year 3)	44	4	
Winter Rapeseed	20th September (year 3)	12	0	0
	30th February (year 4)	32		
	1st April (year 4)	32		
Winter Wheat	10th May (year 5)	40	38.6	1.8

2.3 Animal Growth Model

The Animal Growth Model was developed at SAC (Emmans 1997; Wellock *et al*. 2004; Kyriazakis 2006) and predicts potential growth and voluntary food intake of an animal. The model begins at the day of conception and predicts body composition and feed energy requirement for each day of increasing age. It is assumed that no constraints are acting on the animal. The protein content of an animal's body is calculated using the equation:

$$\mathbf{P = P_m\, e^{-e\,(-G0-B.t)}}$$

where **P** represents the current protein weight (kg), $\mathbf{P_m}$ represents the mature protein weight, which is assumed to be approximately 15% of the mature weight (kg), $\mathbf{G_0}$ represents the Gompertz variable -2.586, which is a measure of the relative protein content of the animal at conception. **B** represents a rate parameter of growth over time (t). Different genotypes can be modelled by adjusting the input parameters **B** and $\mathbf{P_m}$ and average values for modern pig genotypes of $\mathbf{P_m}$ = 40 kg and **B** = 0.01 (Kyriazakis 2006) were selected.

The lipid growth is calculated assuming that the animal is not constrained in anyway, e.g. free from infection. It is also assumed that animals have a particular target fat weight that it aims to achieve under ideal conditions. **Q** is the degree of fatness of the mature animal and is defined as:

$$\mathbf{Q = L_m / P_m}$$

where $\mathbf{L_m}$ is defined as the mature lipid mass of an animal that is achieving its genetic potential for growth and therefore can be related to its protein content using this equation,

$$\mathbf{L = L_m \times (P/P_m)^b}.$$

L is the lipid mass and **P** is current protein mass and **b** is defined as:

$$\mathbf{b = 1.46 \times Q^{\,0.23}.}$$

The body composition of the pig consists of protein, lipid, ash and associated water (Emmans 1997). The quantity of bone ash has been shown to be related directly to the quantity of protein so that for every kg of protein there is 0.2 kg of ash (Kyriazakis 2006). Body water (**BWat**) is also related to the protein weight and is defined as:

$$\mathbf{BWat = 4.889 \times P^{0.855}.}$$

The empty body weight can then be calculated from the sum of the protein, lipid, ash and water weight. Since gut fill is assumed to be 5% of the body weight (Kyriazakis 2006), the body weight is divided by 0.95 to estimate body weight.

Energy required to meet the cost of protein synthesis has been shown to be 53.6 MJ ME/kg (Kyriazakis 2006; Ringel & Susenbeth 2009) of protein retained. There is also a costs for lipid retention of 52.4 MJ ME/kg of lipid deposited (Kyriazakis 2006). To calculate the energy required achieving the potential growth of protein and lipid, the rates of protein and lipid deposition each day is determined. To calculate this, the protein growth equation is differentiated to find the rate of protein growth with respect to time (protein retention):

$$\mathbf{dP/dt = P \times B \times \ln(P_m/P)}$$

where **P** is the current protein mass (kg), **B** is the aforementioned rate parameter and **Pm** is the mature protein weight (kg). For lipid growth with respect to time (lipid retention) the differential equation is:

$$\mathbf{dL/dt = L \times B \times \ln(L_m/L)}$$

Where **L** is the current lipid mass (kg), **B** is the rate parameter and $\mathbf{L_m}$ is the mature lipid mass (kg). The model also determines maintenance energy costs which is related to the protein weight within the body. Maintenance energy costs are expressed as MJ (ME) per day.

$$\mathbf{dM/dt = M_e \times 1.1 \times P^{0.75}}$$

$\mathbf{M_e}$ is a constant energy requirement within a genotype estimated at 1.75 MJ per kg per day (Kyriazakis 2006) and **P** is the protein weight (kg). Only one genotype was considered in the model.

The average daily gains achieved in the Animal Growth Model represent the expected gains of growing pigs within the industry under good management conditions. During the starter phase the average weight gain is predicted as 690g/day and the predicted average gain during the grower and finisher phase is 890g/day.

An important aspect within the Animal Growth Model is the prediction of the amount of N excreted from the pig during the growing period under consideration. The amount of N excreted is dependent on diet composition, feed intake and protein retention. The amount of N which is excreted ($\mathbf{N_{excreted}}$) is calculated by taking the difference between the amount of crude protein intake ($\mathbf{P_{in}}$) and the amount retained by the pig, and then converting this to N by dividing by 6.25 (Salo-vaananen & Koivistoinen 1996; Kyriazakis 2006) to determine the amount of N excreted, and thus is described as:

$$\mathbf{N_{excreted}\ (kg/day) = (P_{in} - dP/dt) / 6.25.}$$

The N excreted is proportioned into an amount in urine and faeces. The amount of N in faeces ($\mathbf{N_{Faeces}}$) was calculated by (Wellock *et al* 2004):

$$\mathbf{N_{Faeces} = CP_{in} \times (1\text{-}D_{il}) + (Pt_{endogenous} \times FR) / 6.25}$$

$\mathbf{CP_{in}}$ represents crude protein intake (kg/day) $\mathbf{D_{il}}$ ileal digestibility (kg/kg) and $\mathbf{Pt_{endogenous}}$ is endogenous protein (unit?). The N excreted in urine ($\mathbf{N_{Urine}}$) is then calculated by:

$$\mathbf{N_{urine} = N_{excreted} - N_{Faeces}}$$

The parameters and values used in the Pig Growth Model are given in Table 17**.**

Table 17 Parameters and values used within the equations of the Animal Growth Model.

	Parameter Value
Protein mass of the mature pig (kg); $\mathbf{P_m}$	40
Gompertz Variable; **Go**	-2.586
Rate of animal growth: **B**	0.01
Mature lipid mass (kg); $\mathbf{L_m}$	80
Metabolic ideal protein (varies with genotype); $\mathbf{m^{ip}}$	0.004
Maintenance energy requirement (MJ/kg$^{0.27}$); $\mathbf{M_e}$	1.75
Water mass calculation;	
Parameter 1	4.889
Parameter 2	0.855
Ash (kg per kg protein)	0.2
Gut fill (% of body weight)	5%
Energetic cost of retaining protein (MJ ME/kg)	53.64
Energetic cost of retaining lipid (MJ ME/kg)	52.4
Activity correction for maintanence energy	1.1

(Emmans 1997b; Kyriazakis 2006)

2.4 Diet Formulation

The diets are formulated as a sub-model in Excel using linear equations. The feed ingredients used in the diets are then incorporated into a crop rotation which would be needed to supply all components of the diets and this crop rotation is modelled in DNDC. The total daily feed requirements (kg/day) for each dietary phase were based on the outputs from the Animal Growth Model and the energy concentration in the diets (MJ ME/day). The additional diet specifications are based upon standards that are currently used in the pig industry and were provided by Alison Johnson (British Quality Pigs, 2008) and given in Table 18.

Table 18 Diet Specifications for starter-finisher pig diets, all data uses minimum specification with the exception of NDF which is given as the maximum.

	Starter	**Grower**	**Finisher**
DE MJ/kg	13.3	12.3	11.6
NDF g/kg	114	150	206
CP g/kg	200	178	173
Total Lysine g/kg	13.5	10.94	9.95
Total M+C g/kg	7.4	6.7	6.49
Total Threonine g/kg	8.2	7.24	6.58
Total Tryptophan g/kg	2.2	2.19	2.04

To calculate the amount of each feed ingredient that was required, the nutritional composition of each feed ingredient was included within the linear model. This included; digestible energy (DE), neutral detergent fibre (NDF) and crude protein (CP). The amino acids lysine, methionine and cysteine, threonine and tryptophan were considered as first, second, third and fourth limiting amino acids. Synthetic forms of these amino acids (SAAs) were into the diets to meet specified requirements. Mass balance was achieved through variation in the amount of barley included. Due to anti-nutritional factors (ANFs) in various diet components, limits were set on ingredients listed in Table 19, which comply with current industry recommendations. The basal diets used in each of the growth phases are given in Tables 20, 21 and 22, which all met the requirements specified in Table 18 and Table 19.

Table 19 Maximum inclusions of ingredients in starter-finisher pig diets.

	Starter	Grower	Finisher
Peas (maximum)		30%	
Beans (maximum)		20%	
Lupins(maximum)		30%	
Soya	Is added if crude protein requirement cannot be achieved.		
Rapeseed meal (maximum)	5 %	10 %	15 %
Wheat feed (maximum)	4 %	15 %	27.5 %
Wheat and barley		No limits	
Molasses	1 %	2 %	3 %
Fat supplement	1.8 %	1.1 %	1 %
Min/vit		2.72%	
SAA		Adjusted to match the requirements	

Table 20 Starter diet composition for each diet scenario.

	Starter (%)			
Ingredient	**Soya**	**Peas**	**Beans**	**Lupins**
Barley	14.1	0	0	0
Wheat	51.3	42.4	53.6	59.5
Peas	-	30.0	-	-
Beans	-	-	20.0	-
Lupins	-	-	-	21.0
Soya	23.0	14.5	14.0	9.0
Rapeseed meal	2.5	5.0	5.0	5.0
Wheat feed	3.9	3.0	2.0	0
Rest	5.2	5.1	5.4	5.5

Table 21 Grower diet composition for each diet scenario.

	Grower (%)			
Ingredient	**Soya**	**Peas**	**Beans**	**Lupins**
Barley	12.3	41.8	7.2	20.9
Wheat	44.3	0	39.2	37
Peas	-	30.0	-	-
Beans	-	-	20.0	-
Lupins	-	-	-	16.9
Soya	14.0	7.5	5.3	3.2
Rapeseed meal	7.0	7.1	10.0	10
Wheat feed	15.0	6.4	10.9	4.5
Rest	7.4	7.2	7.4	7.5

Table 22 Finisher diet composition for each diet scenario.

	Finisher (%)			
Ingredient	**Soya**	**Peas**	**Beans**	**Lupins**
Barley	28.4	15.4	12.3	5.2
Wheat	15.8	6.0	19.0	34.0
Peas	-	30.0	-	-
Beans	-	-	20.0	-
Lupins	-	-	-	12.0
Soya	7	0	0	0
Rapeseed meal	14.0	14.0	14.0	14.0
Wheat feed	27.5	27.5	27.5	27.5
Rest	7.3	7.1	7.2	7.3

2.5 Calculating Emissions from Slurry

2.5.1 Fertilizer Scenarios

From the Animal Growth Model, slurry production per pig place per year is calculated. The environmental impacts associated with slurry were only included in the slurry fertilizer scenario, this includes the environmental impacts from the stored slurry throughout the year. This was calculated from the average amount of slurry stored per day, using a bulk density estimate for slurry of 1.03 kg/m^3 (Lopez-Ridaura *et al*. 2009). The emission equivalent factors are estimated at 5.44 kg CH_4/m^3/year and 0.652 kg NH_3/m^3/year (Lopez-Ridaura *et al*. 2009). An environmental burden is allocated for the amount of slurry exported by lorry and this is assumed to be 1 km from the farm boundaries. However, as the boundary of the system is the farm gate no environmental costs associated with the slurry after it has been exported from the farm boundaries are included.

All calculations of slurry requirements were initially carried out for requirements per hectare (ha) and slurry production was calculated per pig place per day. The total slurry storage was affected by slurry production on the one hand and slurry application on the other. First total slurry requirements (ha) to meet N requirements per crop to produce pig feed ingredients per pig place were calculated for each crop by dividing the total ingredient requirements per pig place by yield per ha. Total yearly slurry requirements were calculated from this acreage and yearly slurry requirements per crop. The total N content of slurry was assumed to be 4.2 kg/T. The amount of slurry available for export per year was calculated as the difference between total slurry production and total slurry requirements for crop growth. Slurry in store was calculated on a daily basis as the cumulative amount of slurry produced minus the amounts applied to crops on the appropriate days. Where this resulted in negative amounts of slurry in store, the amount in store on January 1 was assumed to be equal to the maximum negative amount of slurry in store at any point in the year. Subsequently, the amount of slurry in store was equal to the amount still to be applied to crops later in the year. An environmental cost was assigned to the amount slurry exported, assuming 1km road transport. The actual amount of slurry in store was finally calculated from the amount in store at the beginning of the year, the cumulative amount of slurry produced during the year, the application to specific crops and the export. Using the bulk density estimate for slurry the environmental impacts were calculated.

2.6 Additional Processes

The energy uses for all processes involved within the production system are calculated in the LCA. Data from published literature are used and implemented in the LCA to calculate energy uses within the system. These are further refined to the equivalences to each environmental impact category. The additional processes are associated with the individual models and sub-models used within the LCA framework.

2.6.1 Crop Production

As previously described, crop growth is simulated in DNDC. However the outputs from DNDC only account for crop growth and N inputs into the system. Therefore the energy uses associated with crop management practices are not included in the DNDC simulations. The land management processes included in the LCA are divided into four categories: cultivation, fertilization, spraying and harvesting and the values for fuel energy use were taken from the Defra ISO2050 LCA (Williams *et al*. 2006). The cultivation processes (per ha) include: ploughing (200mm) using a rotary cultivator (4m), spring time harrowing and weeding, using a conventional drill and using Cambridge rolls (Table 23). For each site, the individual soil type affects the energy use required by the machinery for each process. For example a heavy clay soil requires more energy to cultivate than a light loam soil. The total energy required for the cultivation processes for each crop grown at each site therefore depends on the type of soil at the specific site, the energy required for ploughing and harrowing on clay soils were multiplied by 1.7 to account for more energy required to cultivate denser soils (Williams *et al*. 2006). This was applied to more dense East Anglian soils. Fertilizer and slurry application are calculated specifically for each crop, which depends on the number of applications required. The energy required for each process per ha was then converted to the diesel requirement per ha using data from Hansson and Mattson (1999), with a 35.3 MJ equivalent for one litre of diesel.

Table 23 Energy and diesel required per farm operations.

	Total energy MJ/ha	Diesel litres/ha	Kg CO_2 equivalent^{100}/ ha
General cultivation			
Plough (200mm)	1350	38	99.4
Rotary cultivator (4m)	914	26	67.3
spring tine harrows and weeding	300	9	22.1
conventional drill	280	8	20.5
Rolling Cambridge rolls	248	7	18.3
Spraying and fertilising			
Spraying (self propelled)	114	3	8.4
Muck spreader	1259	36	92.7
Disc fertilizer broadcasting	105	3	7.7
Grain Harvesting			
Combine harvester with straw chopping	1134	32	83.5
Grain carting (yield dependent, 8 t/ha)	399	11	29.4

The amount of energy used is then converted to kg CO_2 equivalent^{100}. It is assumed that 1 MJ electricity is equal to 0.261 kg CO_2 equivalent^{100} (Yan 2009) and 1 litre of diesel is equal to 2.6 kg CO_2 equivalent^{100} (Van Belle 2006). Diesel consumption was calculated from the MJ use per ha. The contribution of the initial production of buildings, tractors and other farm machinery to the GWP associated with the production of 1 kg of crop was found to be negligible (Vink *et al.*, 2003) and no further attempts at quantifying these costs were therefore made. To calculate the GWP of the farm operations per diet, the amount of energy required per ha for each process is calculated. This is then converted to GWP per ha and using the output yields from DNDC for each scenario, it is converted to GWP per kg of pig diet ingredient. These values are then multiplied by the amount of ingredient included in the diet.

2.6.1.1 Production of Synthetic Fertilizers

The energy required for production, packaging and transportation of synthetic fertilizers is included for N, P, K fertilizers. The relevant GWP data was taken from the Defra ISO2050 LCA (Williams *et al*. 2006); 6.8 kg CO_2 equivalent^{100}/kg of N fertilizer, 1.2 kg CO_2 equivalent^{100}/kg of P fertilizer and 5.7 kg CO_2 equivalent^{100}/kg of K fertilizer. For each crop, the GWP associated fertilizer applications to 1 ha was divided by crop yield to calculate the GWP associated with all pig diet ingredients. From this data the total GWP associated with fertilizer application.

2.6.1.2 Herbicides and Fungicides

Chemicals are also applied to crops (in both the synthetic and slurry fertilizer scenarios) to prevent weed and fungal growth (Table 24 and Table 25). Although, this data is not required as an input into DNDC, the energy costs associated with their production and application are included in the LCA. Each dose per Ha was allocated 8.0 kg CO_2 equivalent^{100} (Williams *et al* 2006).

Table 24 Herbicide applications.

East Anglia and Yorkshire	Date	Chemical applied	Amount applied (Ha)
Spring Barley	Spring	Metsulfuron-methyl	20g
		Mecoprop-P	1 L
Winter Barley	Autumn	Pendimethalin + flufenacet	2L
	Spring	Mecoprop-P	1 L
Winter Rapeseed	Pre-emergence	Metazachlor	2.5 L
		Fluazifop-P-butyl	0.5 L
Winter Wheat	Autumn	Pendamethalin + picolinafen	2-3 L
	Spring	Fluroxypyr	1 L
Spring Pea	Spring	Terbuthylazine + isoxaben	0.1 L
Spring Bean	Spring	Prosulfacarb	0.5 L
Spring Lupin	Spring	Prosulfacarb	0.5 L

SAC Farm Management Handbook (2009)

Table 25 Fungicide applications.

East Anglia and Yorkshire	Date	Chemical applied	Amount applied (Ha)
Spring Barley	2 applications	Prothioconazole	0.4 L
		Chlorothalonil	1 L
Winter Barley	2 applications	Prothioconazole	0.4 L
		Chlorothalonil	1 L
Winter Rapeseed	Autumn	Prothioconazole	0.3 L
	Spring	Prothioconazole	0.3 L
Winter Wheat	3 applications	Prothioconazole	0.6 L
		Epoxiconazole + chlorothalonil	0.5 L + 1 L
		Strobilurin + triazoles + Chlorothalonil	0.3+0.4+0.75 L
Spring Pea	2 sprays at flowering	Chlorothalonil + azoxystrobin	1 + 0.6 L
Spring Bean	1 spray	Chlorothalonil + tebuconazole	1 + 0.5 L
Spring Lupin	1 spray	Chlorothalonil + tebuconazole	1 + 0.5 L

SAC Farm Management Handbook (2009)

2.6.1.3 Transport

The transport associated with each crop used in the diets is included in the LCA. The assumption was made that the pig farms grow their own crops to produce the pig diets and that crops were transported a return journey of 2 km to the processing and grain storage unit. The GWP associated with transport is 0.000168 kg CO_2 equivalent per kg/km (Dalgaard *et al*. 2007). It is assumed that soya is produced in Brazil and therefore soya transportation from Brazil to the UK is included (for all diet scenarios), assuming 9,980 sea km (and 0.0000106 kg CO_2 equivalent^{100}/km and 850 road km (and 0.000168 kg CO_2 equivalent^{100}/km) (Dalgaard *et al*. 2007).

2.6.1.4 Grain Processing

The grains used in the diets are assumed to be produced on farm and therefore all grain processing also occurs on farm. Energy requirements for grain drying is calculated using published data estimated as 0.26 MJ/kg for cereals, 0.31 MJ/kg for rapeseed and 0.47 MJ/kg for soya (Eriksson *et al*. 2004). The energy costs were then converted to GWP (kg CO_2 equivalent^{100}).

2.6.2 Additional Processes – Pig Housing

The energy included for heating and lighting in the pig house has been included in the analysis. The data was sourced through existing LCAs on pig production and was calculated per kg pig (live weight). The amount of electricity per pig was 53 MJ (Eriksson 2004) which was then converted to GWP using data from Yan (2009), which assumes 1 MJ of energy equals 0.261 kg CO_2 equivalent^{100}. The environmental cost for all impact categories is also included in the production of synthetic amino acids (SAA), which is assumed to be equal to 3.6 kg CO_2 equivalent^{100} per kg SAA. The proportion of the diet called rest consists of molasses, fats, vitamins and minerals which have a GWP of 0.4 kg CO_2 equivalent^{100} per kg (Eriksson 2004).

2.6.3 Additional Energy Costs

Many of the additional costs in terms of GWP (for example costs of producing farm machinery, veterinary costs, etc.) are extremely small, sometimes considered negligible (Vink, 2003). Therefore as no accurate value could be estimated, no additional costs have been included in the current LCA.

2.7 Integration of the Models

To calculate the environmental impacts for the complete system, the outputs from the models and sub-models are integrated in Microsoft Excel.

2.7.1 GWP

First, selected output files from the DNDC models were used to calculate the GWP associated with crop production. The output files from the last five years of the twenty year rotation are used, which allowed soil C and N to stabilise in the first fifteen years of the simulation. From each file, the total days were used from the plant date of the previous crop to the plant date of the crop of interest so that the total life cycle of the system is included. This allowed the soil content and gas emissions to be captured throughout the rotation and allow for fluctuations in GHG emissions from soils. The GWP associated with crop production was estimated on the basis of soil C balance, CO_2 emissions and N_2O-N emissions. The C and N outputs from DNDC were converted to CO_2, CH_4 and N_2O. To convert the change in soil C to CO_2 it was multiplied by 3.67, C was converted to CH_4 by multiplying by 1.33 and N was converted to N_2O by multiplying by 1.57 (all on the basis of molecular weight). IPCC conversion factors are used to convert the GHG to CO_2 equivalent by multiplying CO_2 by 1, CH_4 by 25 and N_2O by 298 (IPCC, 2007). The totals are then summed and this value is the GWP expressed kg CO_2 equivalent^{100} associated with the production per one ha of each crop.

The C yield for each crop is then converted to grain DM yield via division by 0.45 (actual yield is assumed to be 45 % C (Thornley 1998)). To convert DM yield to product ingredient of the pig diets it is divided again by 0.84 to account for moisture content. Some crops are not fed in their entirety, i.e. rapeseed meal, wheat feed and soybean meal are components of rapeseed, wheat and soya, respectively. Within the integration model the GWP of these products are calculated, firstly, calculating the feed ingredient as a proportion of crop yield (i.e. 20 % for wheat feed, 55 % for rape and 80 % for soya). Subsequently, the GWP allocated to pig ingredients was based on the economic value of the crop component, i.e. the economic value of the feed ingredient as fed were estimated (all relative to the grain crop as a whole); 10 % (wheat feed), 30 % (rape) and 70 % (soya) of the GWP of the crop as a whole (Eriksson *et al*. 2004; Thomassen 2008). For example, calculating the GWP of rapeseed meal, firstly the proportion of the rapeseed grain yield used in rapeseed meal is calculated (55 % of grain crop) and secondly using this value to calculate using economic allocation 67 % of the rapeseed crop is used. The GWP which has been calculated per kg diet ingredient can then be applied to calculate the GWP of the total consumed diet by multiplying the amount (kg) of each ingredient by the GWP per kg. Once the GWP has been calculated for the total of all diets, the GWP per kg of pig is calculated by dividing the total GWP by the total weight gain from 12 kg to 105 kg. All additional processes are then included into the integration model.

2.8 Other Environmental Consequences of Pig Production

In addition to GWP, two other major environmental impacts of pig production scenarios have been calculated; eutrophication and acidification.

2.8.1 Eutrophication

The eutrophication potential associated with pig production is calculated for each scenario on the basis of the major contributing substances, which include ammonia (NH_3), nitrate and nitrite (NO_x) and phosphorus (P). Eutrophication is generally expressed in units of PO_4 and equivalency factors are required to translate the contributing substances to this unit. Based on literature data from Misselbrook (2000) and Huijbregts & Seppala (2001), the eutrophication potential per kg pure N was considered equivalent to 0.42 kg PO_4 with corresponding values for NH_3 and NO_x of 0.35 and 0.13 kg PO_4 equivalent respectively. The PO_4 equivalent of 1 kg pure P is based directly on atomic weights, 3.06 kg PO_4. Total eutrophication potential associated with each process is then calculated as the sum of each contributing substance multiplied by its equivalency factor. The above conversion factors were applied to the following parts of the systems. Firstly quantification of fertilizer inputs is made for the synthetic fertilizer scenario which includes N, P and K fertilizers. Data was used from Williams *et al*. (2006) to calculate the eutrophication cost of producing 1 kg of synthetic fertilizer: 1 kg N fertilizer (ammonium nitrate) = 0.0005 kg PO_4, 1 kg P (triple super phosphate) fertilizer = 0.000074 kg PO_4 equivalent, 1 kg K (K as K) fertilizer = 0.00003 kg PO_4 equivalent. Pesticide production was also quantified using data from Williams *et al* (2006) this was assumed to be 0.015 kg PO_4 per dose. The calculations were initially made per ha and fertilizer requirements were then converted to per kg crop by dividing by the total grain yield per hectare. This value was then multiplied by the amount of each crop in the diet for each crop to determine the contribution per pig in each diet scenario.

After fertilizer application, NO_3 and PO_4 are at risk of being leached from the soils. Therefore to quantify the proportion of N leached two different approaches were used. The first method was a simplified approach using literature values assuming that from the applied N fertilizer to each crop per hectare, 4 % is lost as NH_3 (Beusen *et al*. 2008). The second method used output files from the DNDC simulations, calculating the total N, NO and N_2 leached for each crop. This allows for a comparison to be made from DNDC predictions of N leached with the assumption of potential N leached related to N input. Losses from P applications are calculated on a ha basis, using the assumptions that 1 % of applied P fertilizer lost through leaching (Chen *et al*. 2006). The slurry fertilizer scenario is calculated in the same way as the synthetic fertilizer scenario. The storage of slurry is an additional contributor to eutrophication via the release of NH_3. Lopez-Ridaura *et al* (2009) propose that 0.00179 kg of NH_3 is emitted per $1m^3$ of stored slurry per day, therefore this was applied assuming a density of 1030kg of slurry/m^3 (this includes emissions produced from slurry in buildings). The amounts of slurry produced and stored were calculated and from these values the total release of NH_3 from stored slurry was derived.

The production of NO_x from diesel combustion used in farm operations and transport is also included in the calculations to determine eutrophication potential per kg pig. This was calculated using the data in Table 23 for diesel requirements for farm operations, and using data from the Freight Transportation Services (2008) per litre of diesel consumed, which is 0.026 kg NO_x/l. This was (as previously described) calculated on a per ha basis and from this converted to per kg pig. Finally, the release of NO_x from diesel combustion during transportation of crops on farm and also including the transport of exported slurry was assumed to be 0.000032 kg NO_x kg/km (Freight Transportation Services, 2008). It was assumed slurry was exported 1 km from the farm, hence the slurry exported was multiplied by the above value. The eutrophication potential for the whole system is expressed per kg pig, therefore the result for the complete system is divided by the total weight gain of pig to determine the value per kg pig.

2.8.2 Acidification

Contributing substances to acidification include SO_x, NH_3 and NO_x. Acidification is generally expressed as kg SO_4, the equivalency factors are 1.88 kg SO_x/kg NH_3 and 0.7 kg SO_x/kg NO_x and 1kg SO_x/kg SO_x (Cabaraban *et al.* 2008; Thomassen 2008; Basset-Mens *et al.* 2009). The emissions of NH_3 and NO_x associated with crop production and slurry storage are also used to calculate the acidification potential, but using the acidification potential conversion factors stated above. The combustion of diesel required for farm operations releases NO_x and SO_x, which is 0.026 kg and 0.00068 kg per litre of diesel respectively (Freight Transportation Services, 2008). The transport of crops on farm and the transport of exported slurry also contribute to acidification potential. It has been assumed to transport 1 kg per 1 km is 0.000032 kg of NO_x, and in addition 0.0000013 kg of SO_x is released. The production of pesticides and fertilizers also contribute to the acidification potential; 1kg of pesticide = 0.096 kg SO_4 equivalent, 1 kg N (ammonium nitrate) fertilizer = 0.0047 kg SO_4 equivalent, 1 kg P (triple super phosphate) fertilizer = 0.008 kg SO_4 equivalent and 1 kg K (K as K) fertilizer = 0.0047 kg SO_4 equivalent 1 kg. Finally, the acidification potential to produce SAA required in the diets equals 0.041 kg SO_4 equivalent per kg. All pesticides and fertilizers were initially calculated on a per hectare basis using the amount applied to each hectare for each crop (in each scenario). This was then converted to per kg of feed ingredient by dividing by the total yield and then multiplied by the amount of each feed. These values were then divided by the total weight gain and multiplied by the acidification equivalency factor for each contributing substance to calculate the total acidification potential of producing 1 kg pig live weight.

The described models were then simulated for each scenario. Results were determined for each component of the LCA and then integrated to predict the environmental impacts per kg pig live weight. The results of the model simulations will be given in Chapter III.

CHAPTER III

3 Results

3.1 Model Performance in Terms of Production

3.1.1 DNDC Grain Yield Predictions and Associated GWP

The average grain yields for each crop grown (kg/ha) are given in Figure 3 and Figure 4. The crop yields correspond to the expected average yields for the UK and Brazil (SAC Farm Management Handbook, 2009; Da Silva, 2009).

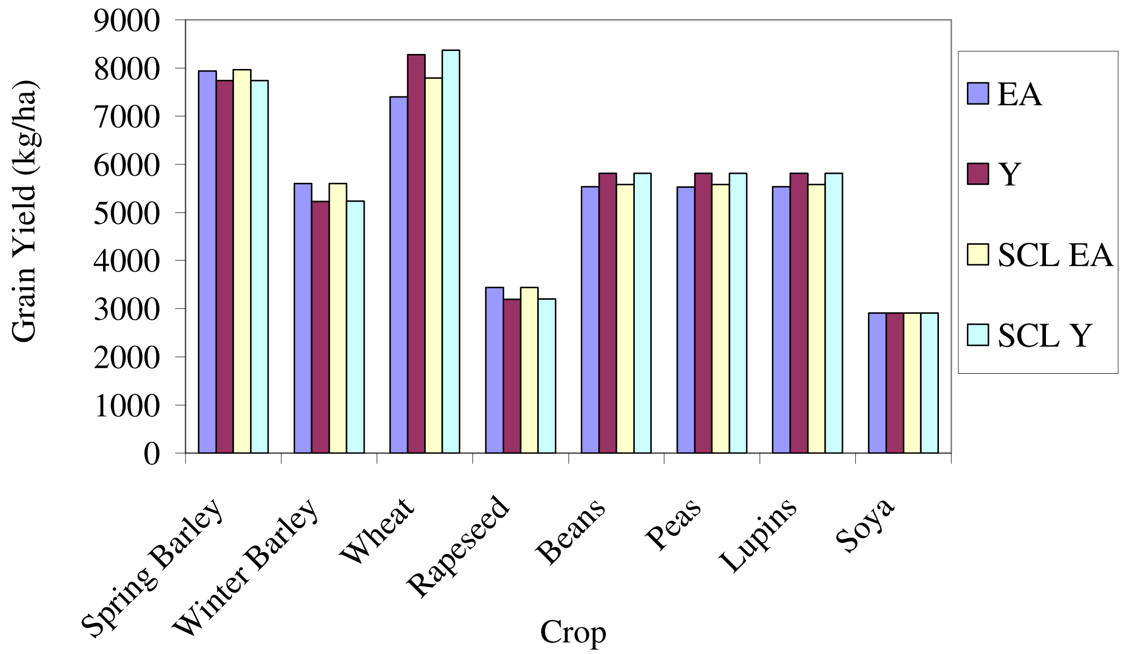

Figure 3 Grain yields for each crop at each site in the synthetic fertilizer scenario.

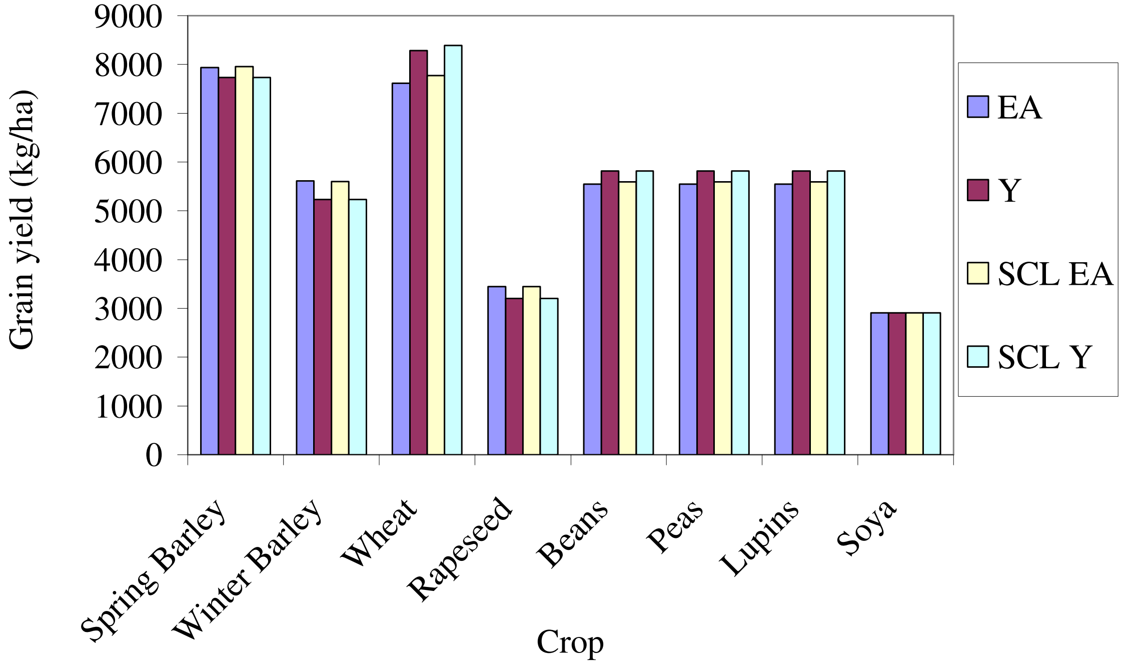

Figure 4 Grain yields for each crop at each site in the slurry fertilizer scenario.

The GHGs produced from crop growth in DNDC simulations at each site are given in Figure 5 to Figure 8 for the synthetic fertilizer scenario. The corresponding emissions for the slurry fertilizer scenario are given in Figure 9 to Figure 12. The GHGs are given as CO_2 and N_2O. The CH_4 emissions are negligible during crop growth and have therefore been emitted from the graphs. The GHGs are the average of the rotations at each site. The GHGs for soya growth are from the Brazilian rotation in each Figure.

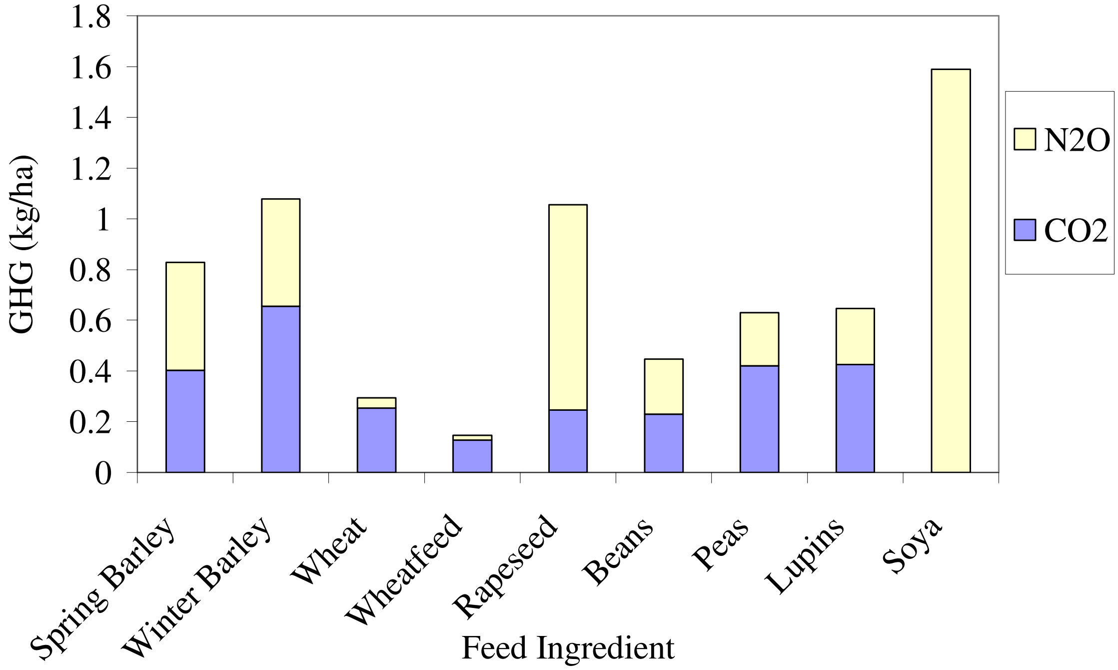

Figure 5 GHGs produced from each feed ingredient grown in East Anglia in the synthetic fertilizer scenario.

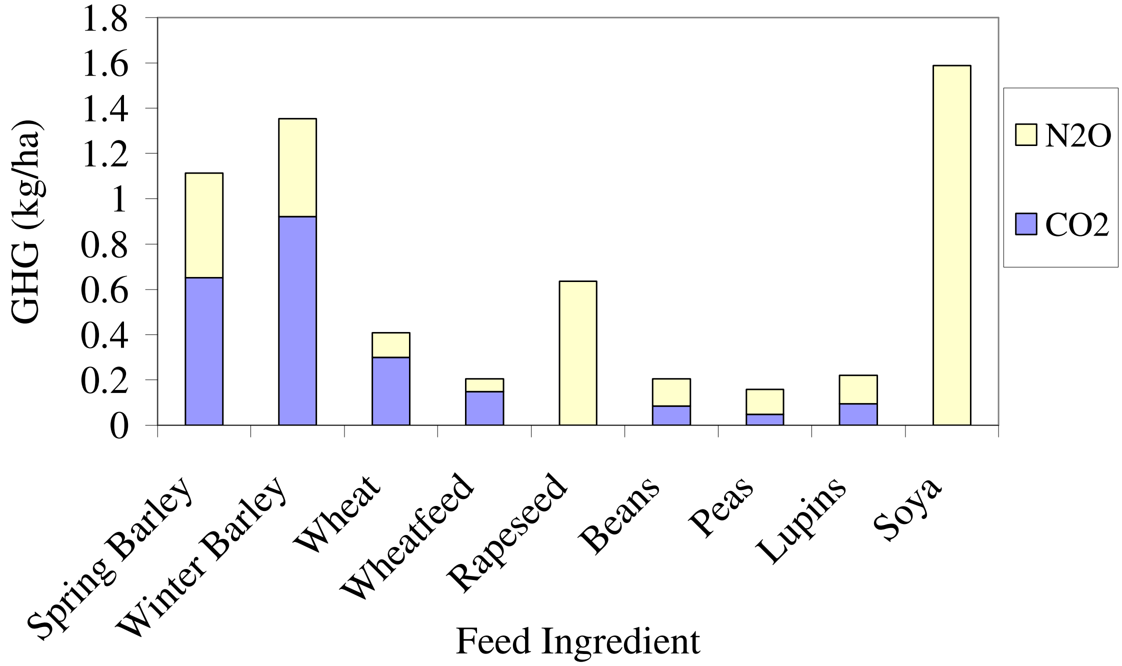

Figure 6 GHGs produced from each feed ingredient grown in Yorkshire in the synthetic fertilizer scenario.

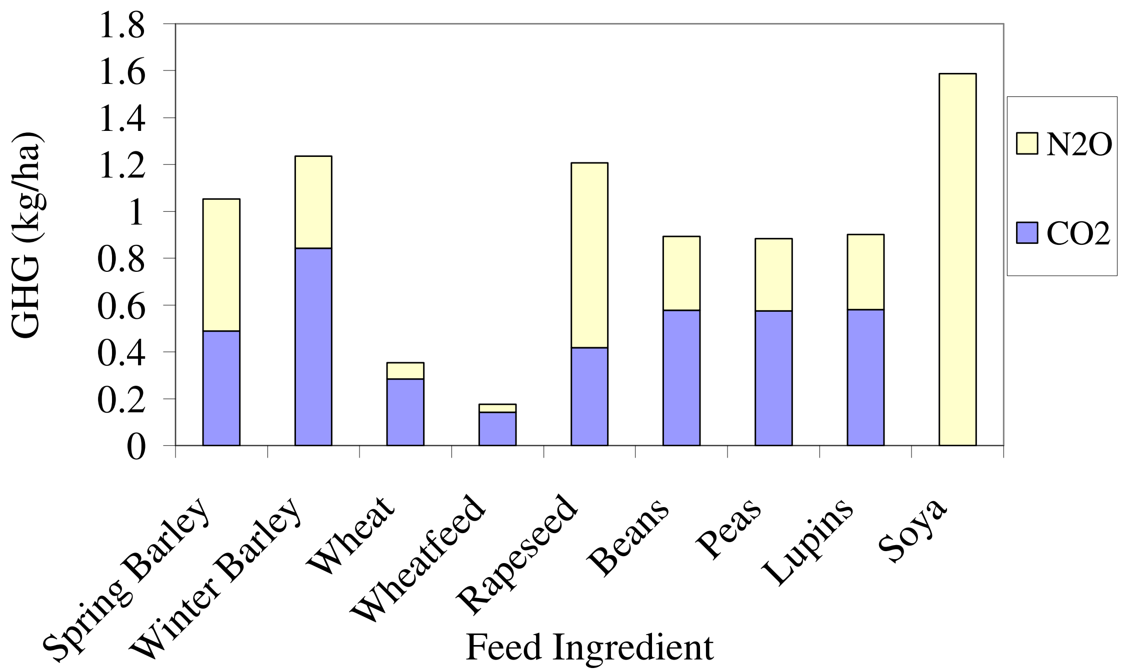

Figure 7 GHGs produced from each feed ingredient grown for silty clay loam in East Anglia in the synthetic fertilizer scenario.

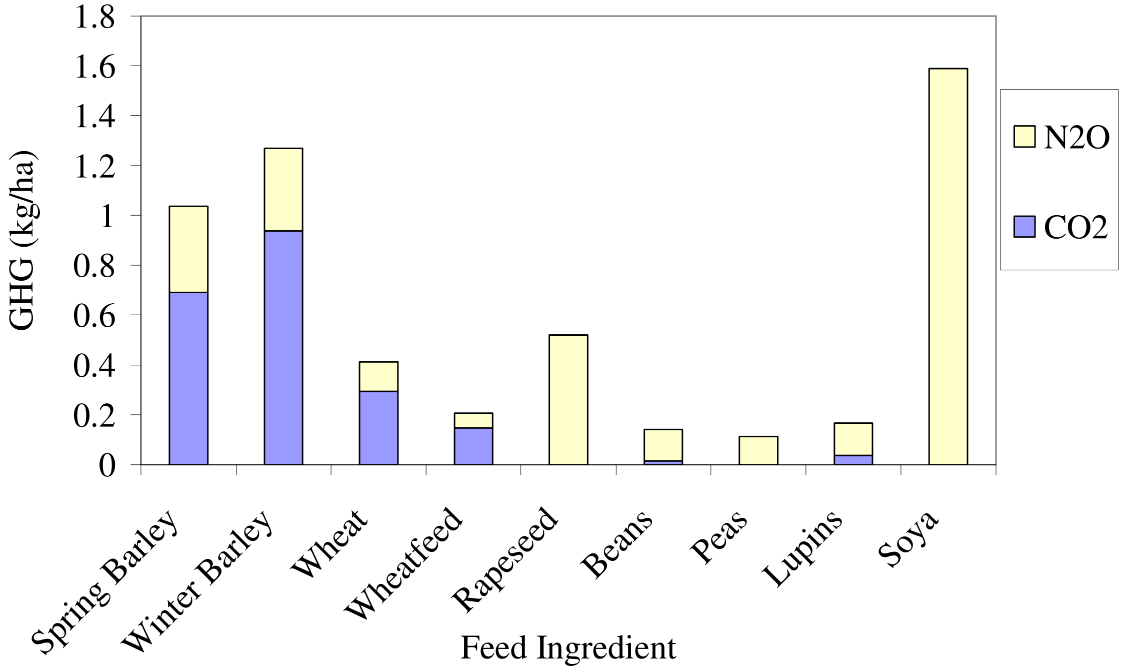

Figure 8 GHGs produced from each feed ingredient grown for silty clay loam in Yorkshire in the synthetic fertilizer scenario.

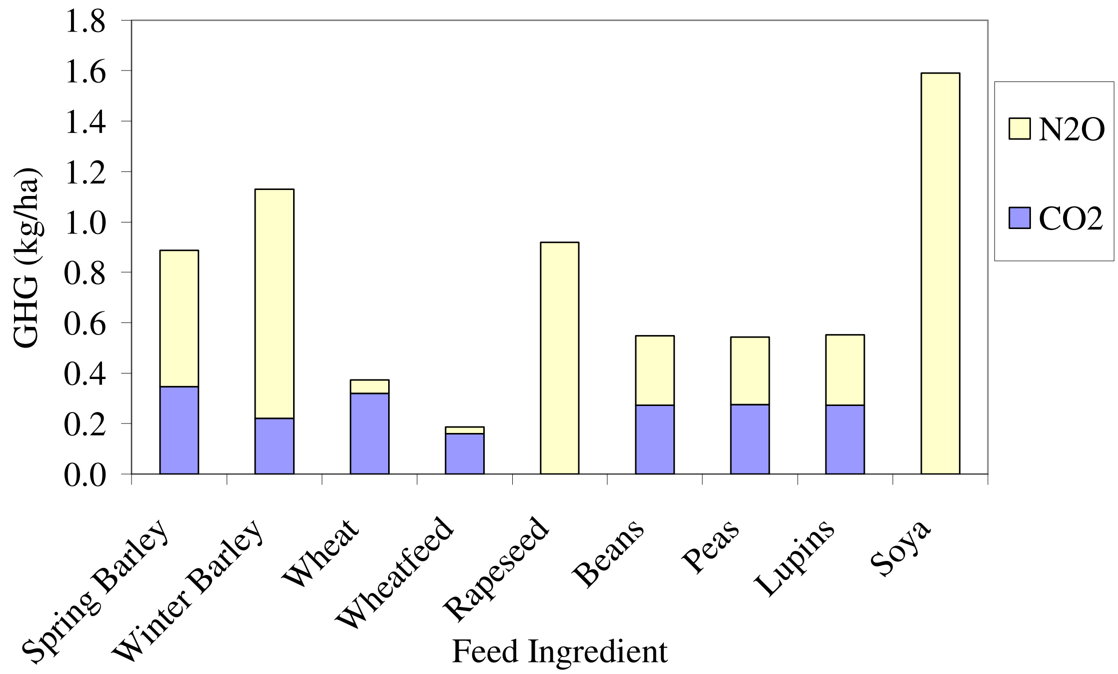

Figure 9 GHGs produced from each feed ingredient grown in East Anglia in the slurry fertilizer scenario.

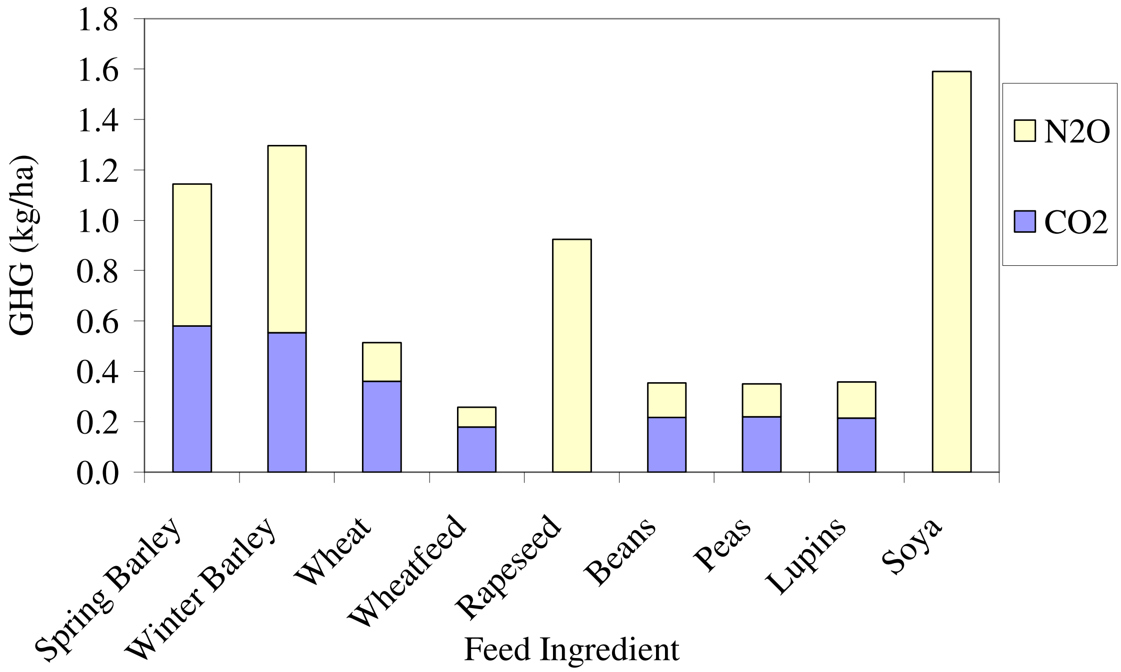

Figure 10 GHGs produced from each feed ingredient grown in Yorkshire in the slurry fertilizer scenario.

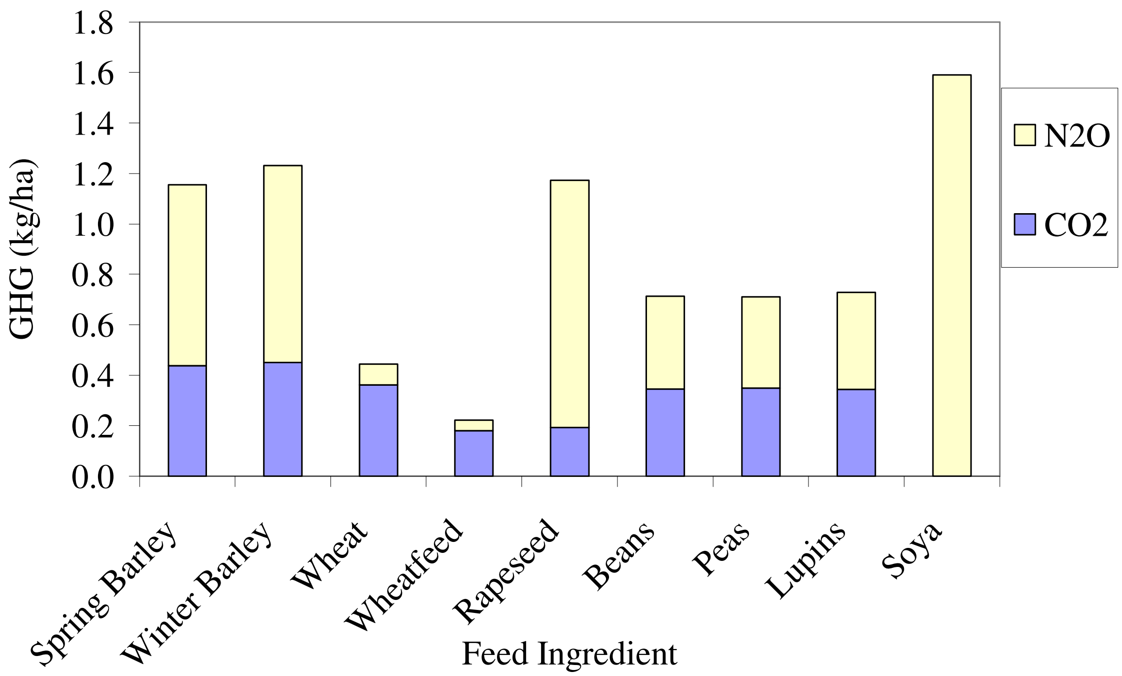

Figure 11 GHGs produced from each feed ingredient grown for silty clay loam in East Anglia in the slurry fertilizer scenario.

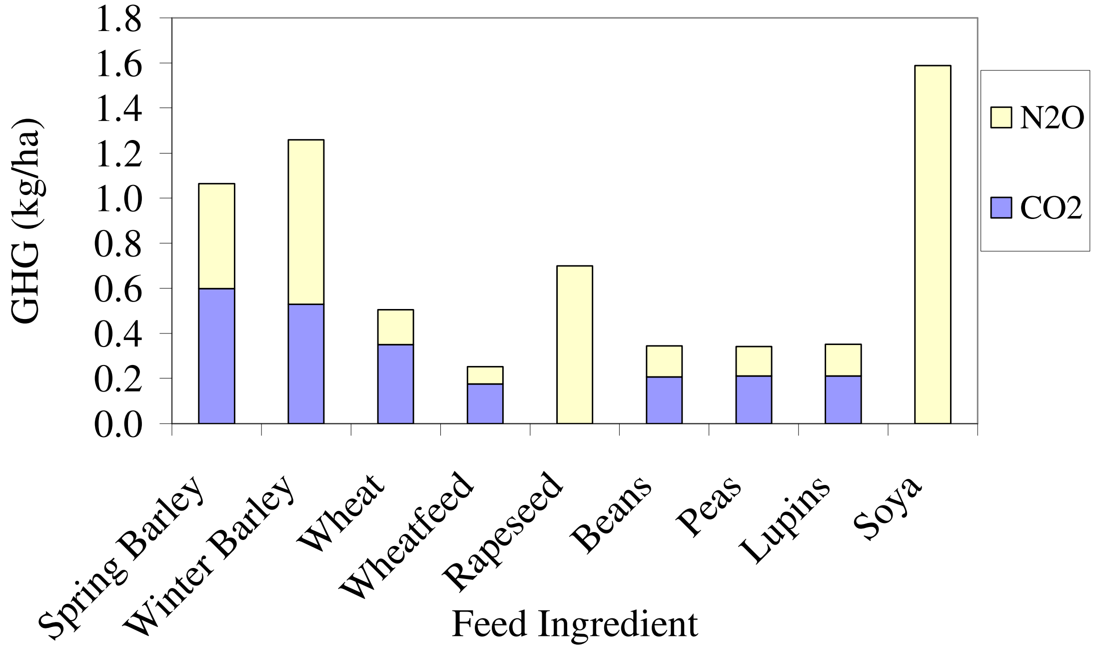

Figure 12 GHGs produced from each feed ingredient grown for silty clay loam in Yorkshire in the slurry fertilizer scenario.

The GHG emissions per feed ingredient are generally higher at each site in the slurry fertilizer scenario when compared with the synthetic fertilizer scenario. The total GWP per kg legume in the slurry fertilizer scenario range from 0.34 to 0.71 kg CO_2 equivalent^{100} per kg beans, 0.34 to 0.71 kg CO_2 equivalent^{100} per kg peas and 0.35 to 0.73 kg CO_2 equivalent^{100} per kg lupins. Compared with the synthetic fertilizer scenario, the total GWP per kg beans ranges from 0.14 to 0.89 kg CO_2 equivalent^{100}, 0.10 to 0.88 kg CO_2 equivalent^{100} per kg peas and 0.17 to 0.90 kg CO_2 equivalent^{100} per kg lupins. Generally, lupins produce the highest N_2O and the lowest CO_2 emissions and overall peas produce the lowest amount of both N_2O and CO_2 emissions. The large variation between the GWP that occurs between sites is due to the different combinations of both climate and soil type in the DNDC simulations.

There are also appreciably more GHGs emitted from winter barley crops grown at all sites in both fertilizer scenarios. In the slurry fertilizer scenario at all sites higher N_2O

emissions occurred from winter barley, however in the synthetic fertilizer scenario more CO_2 emissions occurred from winter barley. This indicates that when slurry is applied to crops there are increased levels of N_2O. This is also the case for rapeseed and in some scenarios from the DNDC simulations only N_2O emissions occurred. The GHG emissions for soya growth are independent from the UK rotations and only N_2O emissions occurred. Percentage contributions of each GHG to total kg CO_2 equivalent^{100} are given in Appendix B.

The GWP associated with the growth of 1 kg feed ingredient for all sites and diet scenarios, including the synthetic and slurry fertilizer scenarios are given in Table 26 to Table **29**. The results are direct outputs of GHGs from DNDC which are converted to kg CO_2 equivalent^{100} and the environmental impacts of crop processing are also included. The GWP for each cereal crop is the average of each site from the bean, pea and lupin rotations.

Table 26 The GWP (kg CO_2 equivalent^{100}) per kg feed ingredient at East Anglia for both fertilizer scenarios.

	East Anglia	
	Synthetic fertilizer scenario	**Slurry fertilizer scenario**
Beans	0.54	0.65
Pea	0.73	0.64
Lupin	0.74	0.65
Spring barley	0.98	0.98
Winter barley	1.35	1.27
Wheat	0.48	0.49
Wheatfeed	0.17	0.21
Rapeseed	1.58	1.29
Soya	2.35	2.35

Table 27 The GWP (kg CO_2 equivalent^{100}) per kg feed ingredient at Yorkshire for both fertilizer scenarios.

Yorkshire		
	Synthetic fertilizer scenario	**Slurry fertilizer scenario**
Beans	0.30	0.45
Pea	0.25	0.46
Lupin	0.31	0.45
Spring barley	1.27	1.24
Winter barley	1.68	1.48
Wheat	0.58	0.62
Wheatfeed	0.23	0.28
Rapeseed	1.15	1.08
Soya	2.35	2.35

Table 28 The GWP (kg CO_2 equivalent^{100}) per kg feed ingredient at silty clay loam Anglia for both fertilizer scenarios.

Silty Clay Loam East Anglia		
	Synthetic fertilizer scenario	**Slurry fertilizer scenario**
Beans	0.99	0.81
Pea	0.98	0.81
Lupin	1.00	0.83
Spring barley	1.20	1.25
Winter barley	1.51	1.37
Wheat	0.53	0.56
Wheatfeed	0.20	0.25
Rapeseed	1.73	1.55
Soya	2.35	2.35

Table 29 The GWP (kg CO_2 equivalent^{100}) per kg feed ingredient at silty clay loam Yorkshire for both fertilizer scenarios.

	Silty Clay Loam Yorkshire	
	Synthetic fertilizer scenario	**Slurry fertilizer scenario**
Beans	0.23	0.44
Pea	0.20	0.43
Lupin	0.26	0.44
Spring barley	1.19	1.16
Winter barley	1.59	1.44
Wheat	0.58	0.61
Wheatfeed	0.23	0.28
Rapeseed	1.04	0.83
Soya	2.35	2.35

Overall, the crops which require more processing have the highest GWP per kg feed ingredient; this includes rapeseed meal and soybean meal. However, this is not consistent at all sites and variations do occur depending on the site conditions. Soya has the highest GWP out all feed ingredients. This is caused by the high GHG emissions during crop growth but also considerably higher amounts of energy required during the extraction process. In all scenarios spring barley and winter barley have relatively high GWP per kg feed ingredient when compared with other feed ingredients caused by higher GHG emissions from DNDC simulations. The GWP for winter barley is highest at both Yorkshire sites in the synthetic fertilizer scenario, 1.59 and 1.68 kg CO_2 equivalent^{100} and 1.44 and 1.48 kg CO_2 equivalent^{100} in the slurry fertilizer scenario. In comparison, winter wheat has a relatively low GWP ranging from 0.48 to 0.58 kg CO_2 equivalent^{100} in the synthetic fertilizer scenario and 0.49 to 0.62 kg CO_2 equivalent^{100} in the slurry fertilizer scenario. Winter wheat grown in East Anglia results has lower GWP when compared with winter wheat grown in Yorkshire.

With regards to the UK grown legumes, when considering all scenarios, lupins grown at silty clay loam East Anglia in the synthetic fertilizer scenario resulted in the highest

GWP 1.00 kg CO_2 equivalent^{100} per kg feed ingredient. In general, legumes grown at the Yorkshire sites resulted in the lowest GWP per kg feed ingredient when compared with the East Anglian sites.

The GWP associated with soya production in Brazil is independent of UK conditions and the GWP per kg soybean meal was 2.35 kg CO_2 equivalent^{100}. From the DNDC simulations all GHG emissions occurred from N_2O (1.59 kg CO_2 equivalent^{100}).

The crop processes, pesticide and fertilizer production contributing to the total GWP for each feed ingredient for all sites are shown in Figure 13 to Figure 16 for the synthetic fertilizer scenarios and in Figure 17 to Figure 20 for the slurry fertilizer scenario. The contribution of each feed ingredient is calculated as the average of the bean, pea and lupin rotation. The Brazil rotation was used to calculate the contribution of soya production to total GWP.

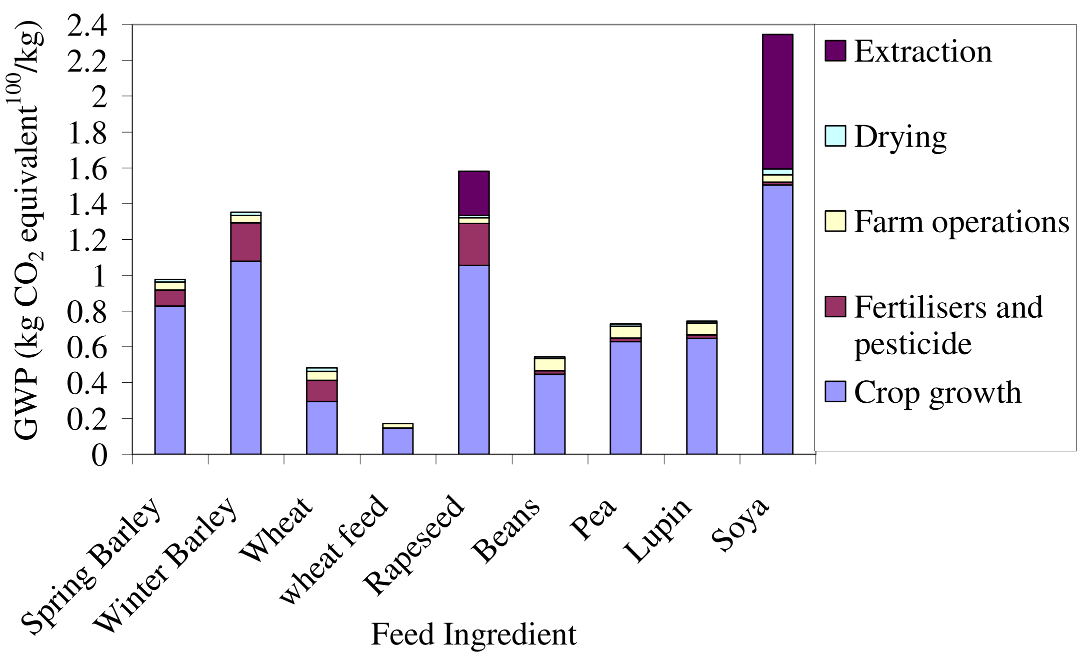

Figure 13 The contribution of crop processes to the total GWP per kg feed ingredient in East Anglia for the synthetic fertilizers scenario.

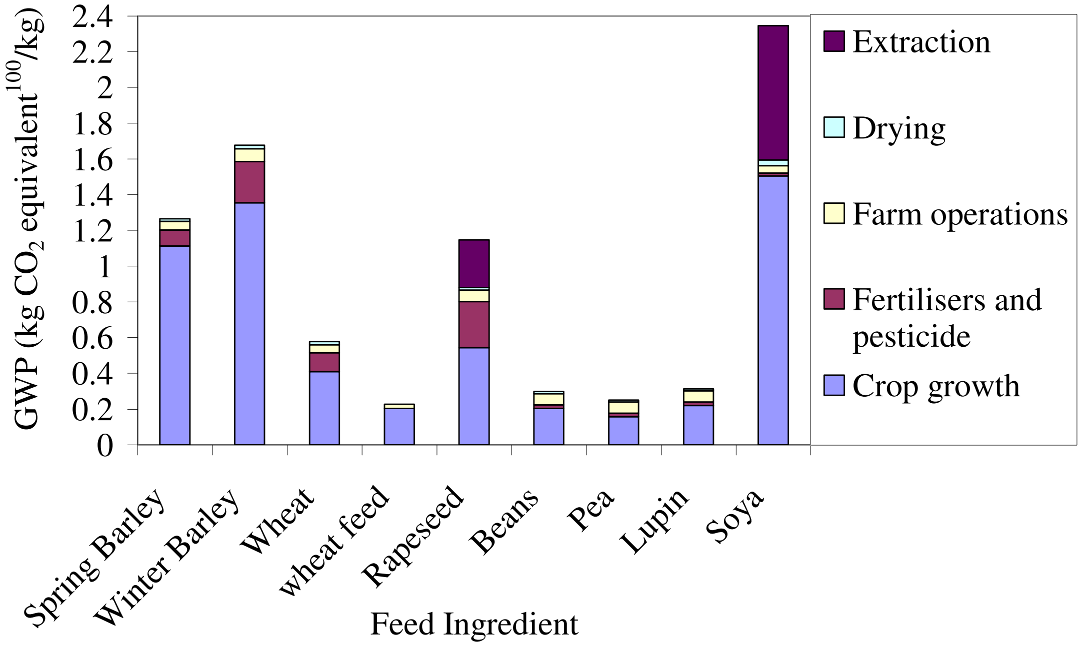

Figure 14 The contribution of crop processes to the total GWP per kg feed ingredient in Yorkshire for the synthetic fertilizers scenario.

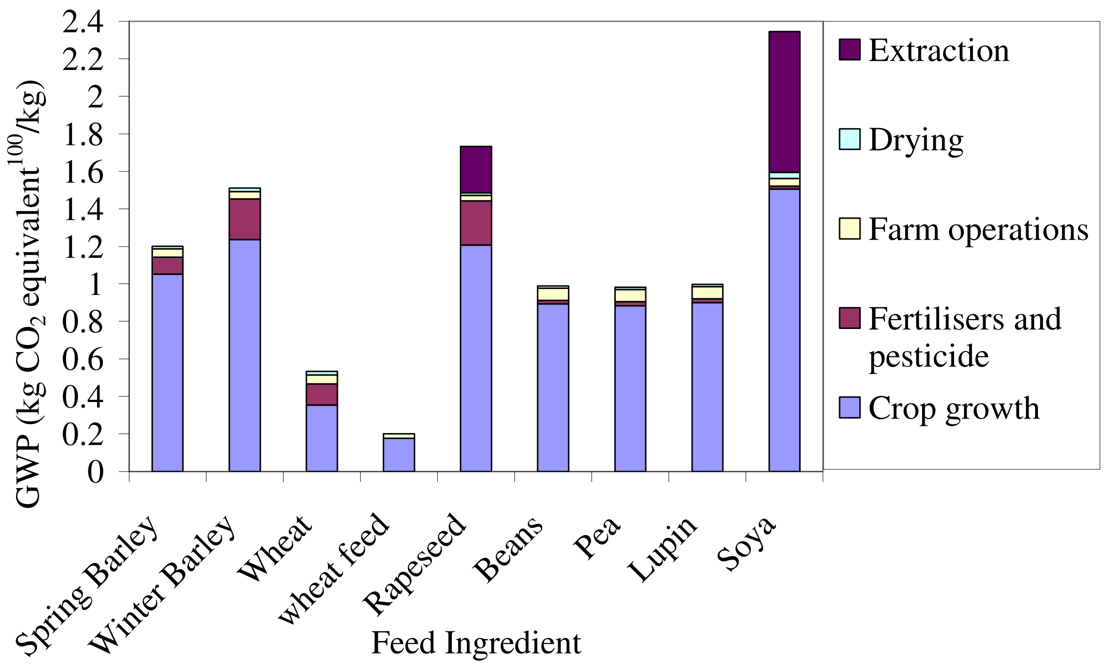

Figure 15 The contribution of crop processes to the total GWP per kg feed ingredient in silty clay loam East Anglia for the synthetic fertilizers scenario.

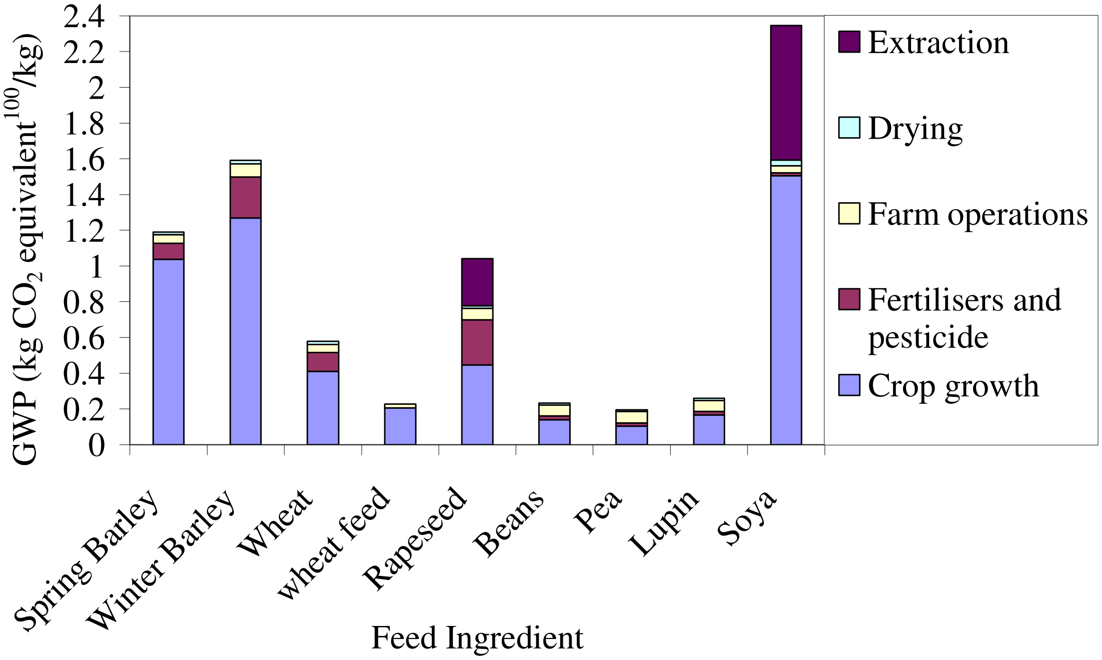

Figure 16 The contribution of crop processes to the total GWP per kg feed ingredient in silty clay loam Yorkshire for the synthetic fertilizers scenario.

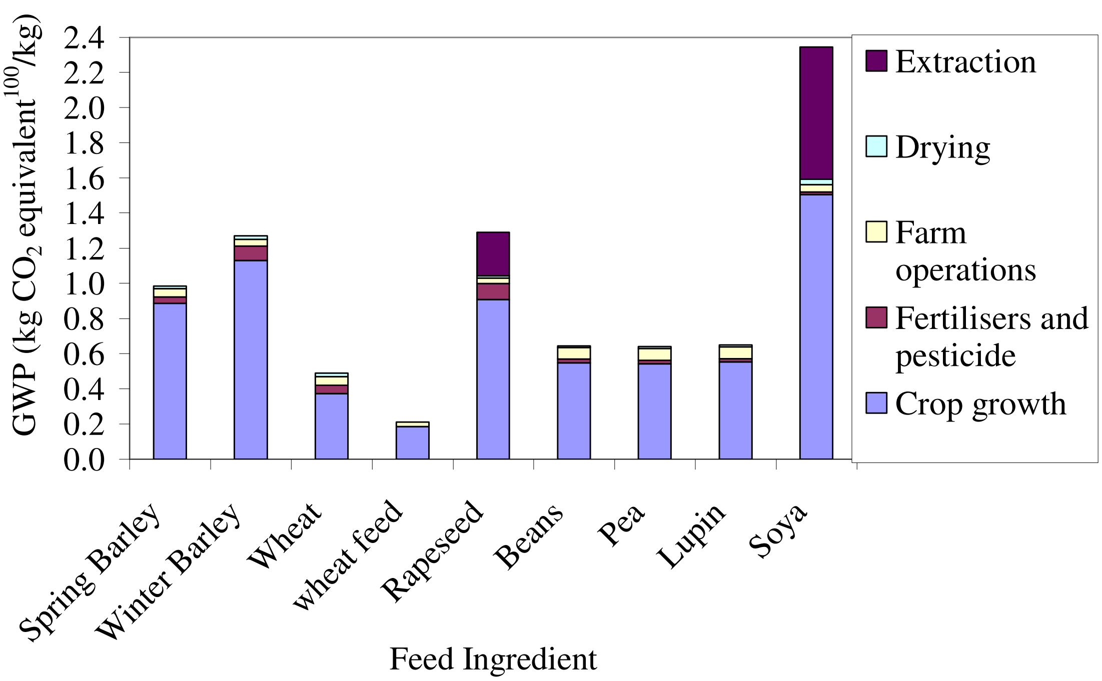

Figure 17 The contribution of crop processes to the total GWP per kg feed ingredient in East Anglia for the slurry fertilizer scenario.

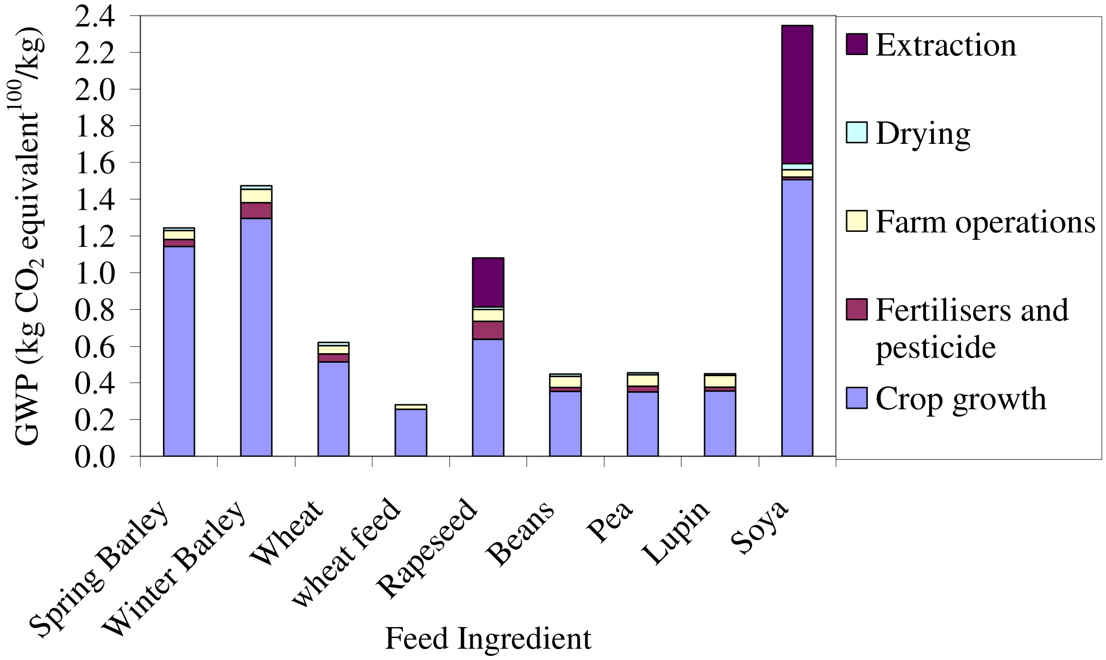

Figure 18 The contribution of crop processes to the total GWP per kg feed ingredient in Yorkshire for the slurry fertilizer scenario.

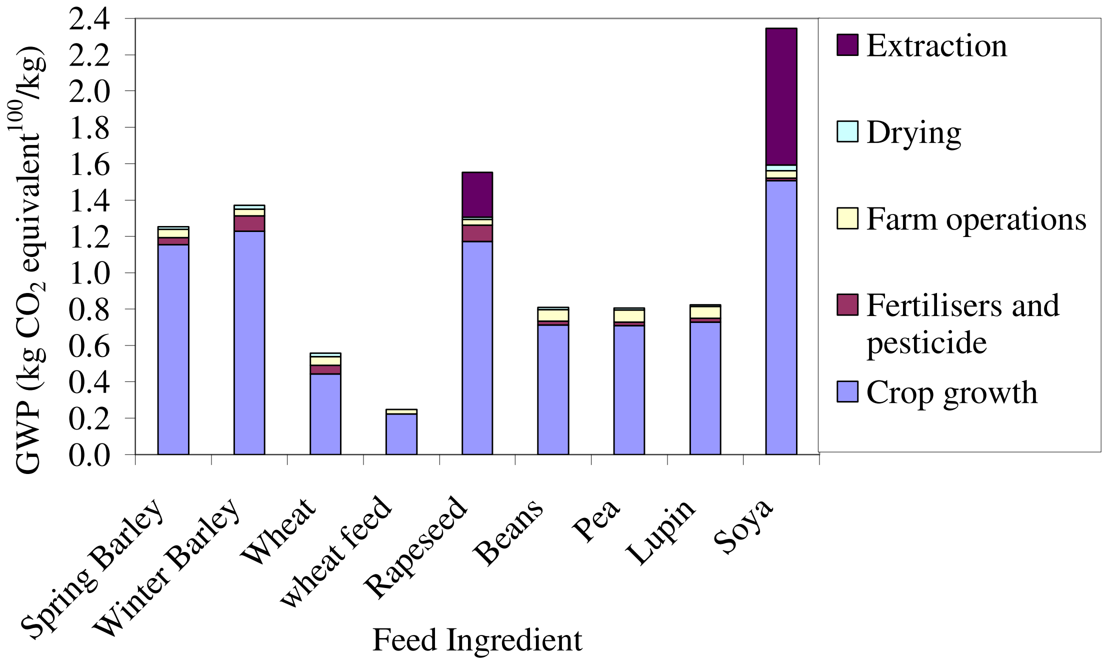

Figure 19 The contribution of crop processes to the total GWP per kg feed ingredient in silty clay loam East Anglia for the slurry fertilizer scenario.

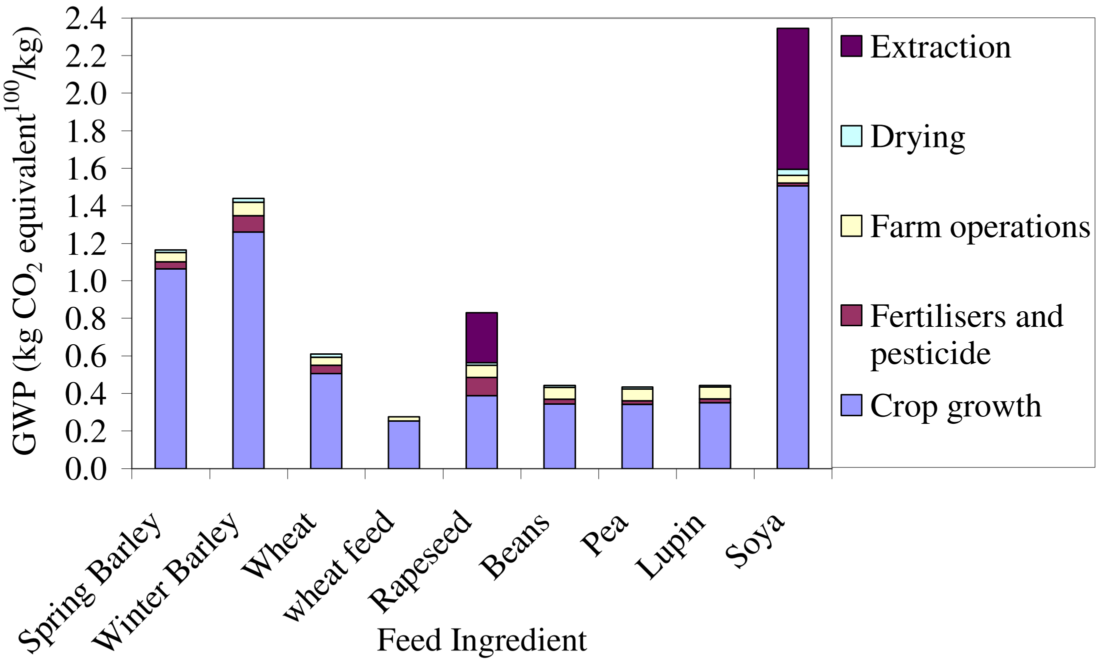

Figure 20 The contribution of crop processes to the total GWP per kg feed ingredient in silty clay loam Yorkshire for the slurry fertilizer scenario.

For all scenarios the highest contribution to GWP per kg feed ingredient is from crop growth (DNDC outputs). Extraction processes are only required to produce rapeseed meal and soybean meal which contributes significantly to the total GWP per kg feed ingredient. The extraction process for rapeseed meal and soybean meal is 0.25 and 0.75 kg CO_2 equivalent^{100} respectively. Grain drying is a relatively low contributor to the GWP per feed ingredient, for all UK crops this ranges between 0.01 to 0.02 kg CO_2 equivalent^{100}, however for Brazilian soya this was higher 0.03 kg CO_2 equivalent^{100}.

The contribution to total GWP from farm operations for each feed ingredient is relatively constant between scenarios. The GWP of farm operations is allocated per Ha. Therefore when considering the impact per kg feed ingredient, the high yielding crops result with a lower impact per kg feed ingredient when compared with low yielding crops. Consequently, the contribution of farm operations to GWP per kg legume ranges from 0.62 to 0.66 kg CO_2 equivalent^{100} whereas for spring barley this ranges from 0.46

to 0.48 kg CO_2 equivalent^{100} for both fertilizer scenarios. The contribution to total GWP from fertilizer and pesticide production varies between crops. This is dependent on the fertilizer requirements of the crop and also the crop yield. In the slurry fertilizer scenario, the contribution is relatively low as 60 % of the N requirement is supplied from slurry, however some synthetic fertilizers are still applied along with pesticides. The contribution for UK cereal crops ranges between 0.037 to 0.096 kg CO_2 equivalent^{100}. Whereas the contribution to the total GWP for the legume crops ranges from 0.020 to 0.021 kg CO_2 equivalent^{100}. However in the synthetic fertilizer scenario, the contribution from fertilizer and pesticide production to total GWP is higher, for UK cereal crops this ranges from 0.08 to 0.26 kg CO_2 equivalent^{100}, as winter barley has the highest fertilizer requirement.

3.1.2 Animal Growth Model Predictions

The predictions from the Animal Growth model are shown in Figure 21 to Figure 24. Figure 21 gives the live weight gain during the grower finisher period from 12 kg (day 36 of age) to 105kg (day 169 of age), respectively. Figure 22 shows the daily live weight gain during the grower finisher period, the total and live weight gain corresponds to industry expectations for a well managed system (Kyriazakis *et al* 2006). Figure 23 shows the daily feed intake per day and Figure 24 shows the daily total N excretion and also N excretion in faeces and urine.

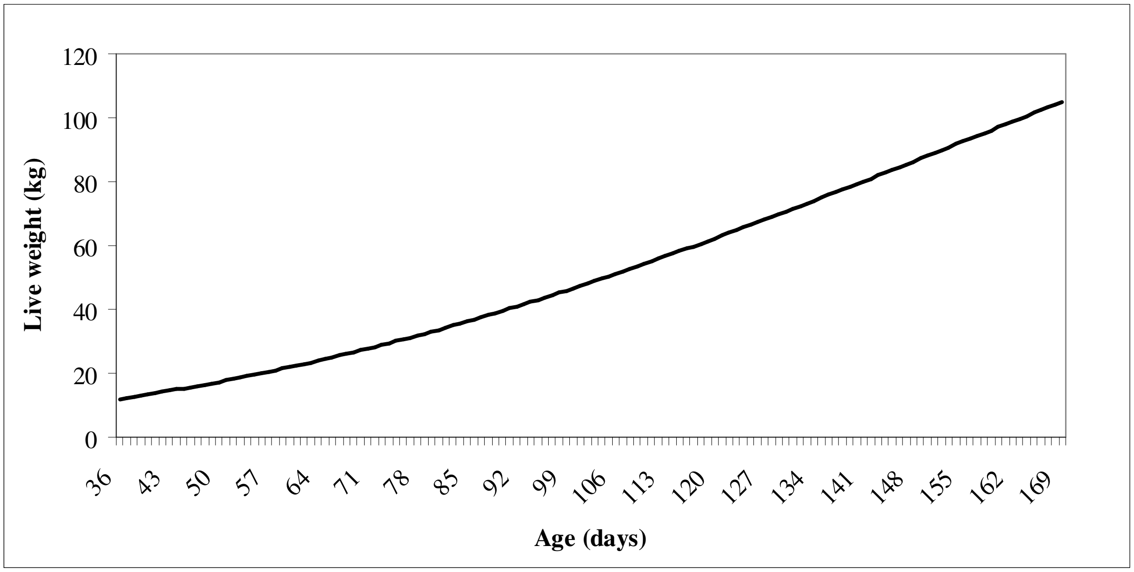

Figure 21 The live weights of the pig during the grower finisher period.

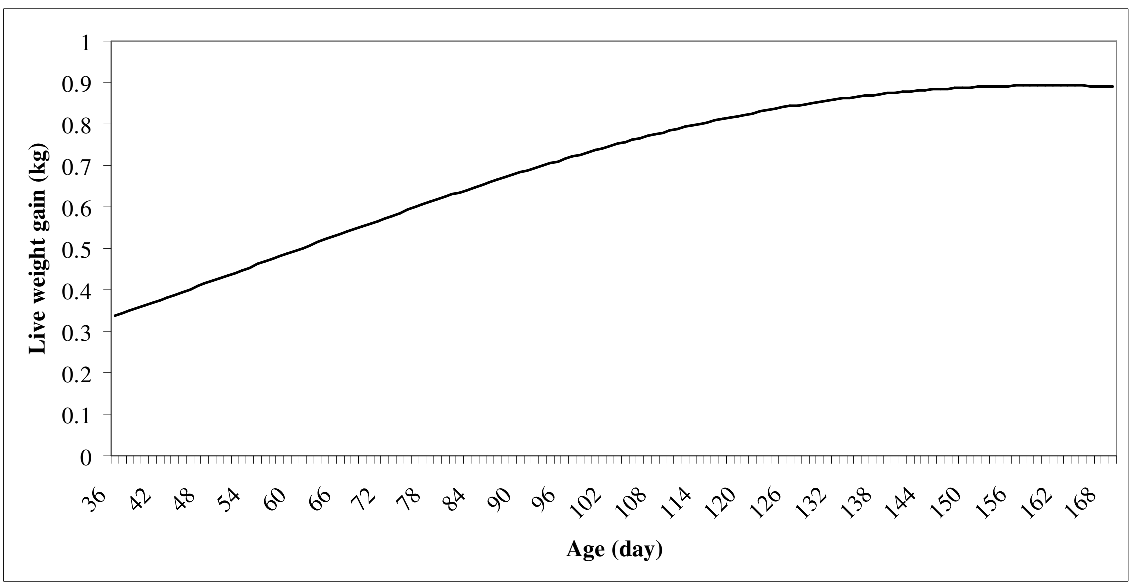

Figure 22 Daily live weight gains of the grower finisher pig.

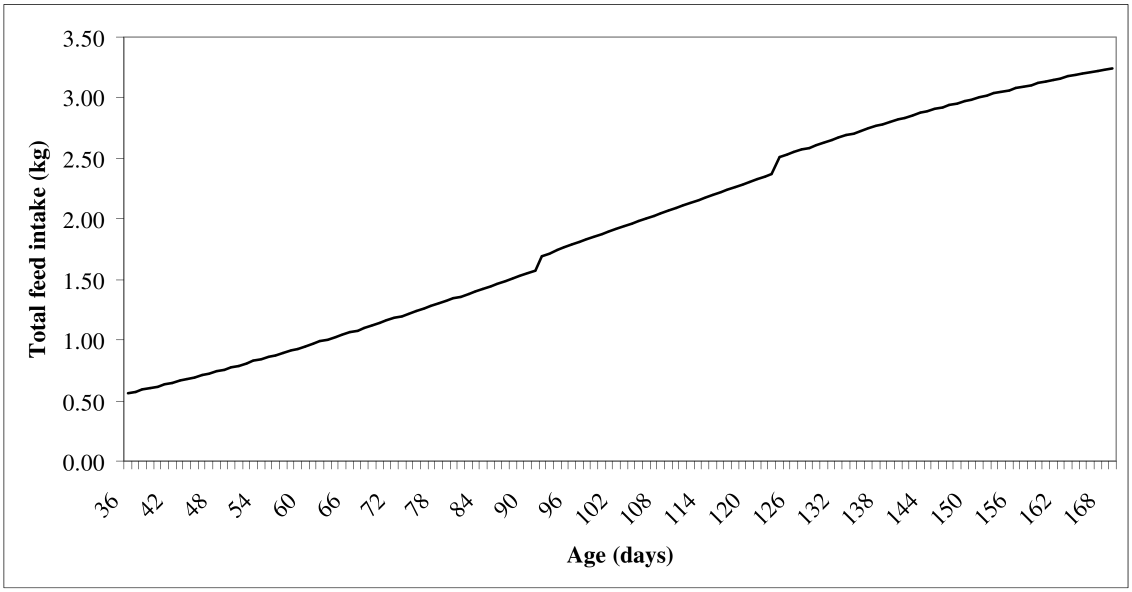

Figure 23 Daily feed intake per pig to achieve the potential weight gain.

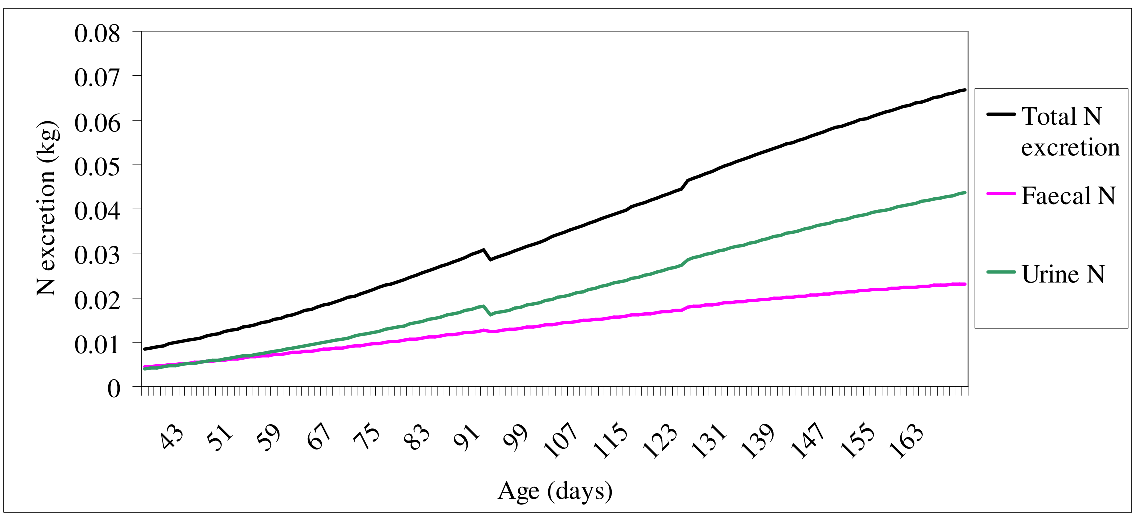

Figure 24 Daily N excretion (kg/day) during the grower finisher period

The breaks in the line indicate when the next diet phase is introduced, the starter diet is fed from 12 kg (36 days of age), at 40 kg (93 days of age) the grower diet is introduced and at 65 kg (124 days of age) the finisher diet is introduced. When the new diet is introduced, this causes a short period of decreased feed intake until the pig adjusts to the new diet. From the total N excreted the proportion of N in urine and faeces is calculated, which gradually increases as the pig ages.

3.2 Environmental Impacts of Pig Production

3.2.1 GWP Associated with Pig Production

The total GWP (kg CO_2 equivalent^{100}) to produce 1 kg of pig live weight is given in Table 30. The results are shown for each diet and site scenario and for each fertilizer scenario. For each diet and site scenario the contributions to total GWP of producing 1 kg pig live weight including all processes within the system are show in Figure 25 to Figure 28 for the synthetic fertilizer scenario and Figure 29 to Figure 32 for the slurry fertilizer scenario.

Table 30 The GWP (kg CO_2 equivalent^{100}) per 1 kg pig (live weight) for each diet and site scenario.

Site and diet scenario	Synthetic fertilizer scenario (kg CO_2 equivalent^{100} per kg pig)	Slurry fertilizer scenario (kg CO_2 equivalent^{100} per kg pig)
East Anglia Beans	2.03	2.40
East Anglia Peas	2.47	2.63
East Anglia Lupins	2.22	2.45
East Anglia Soya	2.52	2.80
Yorkshire Beans	1.93	2.45
Yorkshire Peas	2.18	2.66
Yorkshire Lupins	1.99	2.47
Yorkshire Soya	2.67	3.00
Silty clay loam in East Anglia Beans	2.43	2.67
Silty clay loam in East Anglia Peas	2.86	3.07
Silty clay loam in East Anglia Lupins	2.54	2.79
Silty clay loam in East Anglia Soya	2.73	3.08
Silty clay loam in Yorkshire Beans	1.85	2.29
Silty clay loam in Yorkshire Peas	2.07	2.47
Silty clay loam in Yorkshire Lupins	1.92	2.41
Silty clay loam in Yorkshire Soya	2.61	2.95

The total GWP per kg pig live weight is dependent on the individual conditions at each site, thus causing variations between results. When comparisons are made between fertilizer scenarios the GWPs per kg pig is higher in the slurry fertilizer scenarios at all sites. The soya based diets have the highest GWP per kg pig; in the synthetic fertilizer scenario this ranged from 2.52 kg CO_2 equivalent^{100} (East Anglia) to 2.73 kg CO_2 equivalent^{100} (silty clay loam in East Anglia). With the exception at silty clay loam East Anglia in the synthetic fertilizer scenario where the pea based diets results in the highest GWP per kg pig, 2.86 kg CO_2 equivalent^{100} whereas the GWP per kg pig for the soya based diet this is 2.74 kg CO_2 equivalent^{100}. Overall, the bean based diets have the lowest GWP per kg pig, the lowest at silty clay loam Yorkshire in the synthetic fertilizer scenario, 1.85 kg CO_2 equivalent^{100}.

In the slurry fertilizer scenario the total GWP for the lupin based diets ranged from 2.80 kg CO_2 equivalent^{100} (East Anglia) to 3.08 kg CO_2 equivalent^{100} (silty clay loam East Anglia).

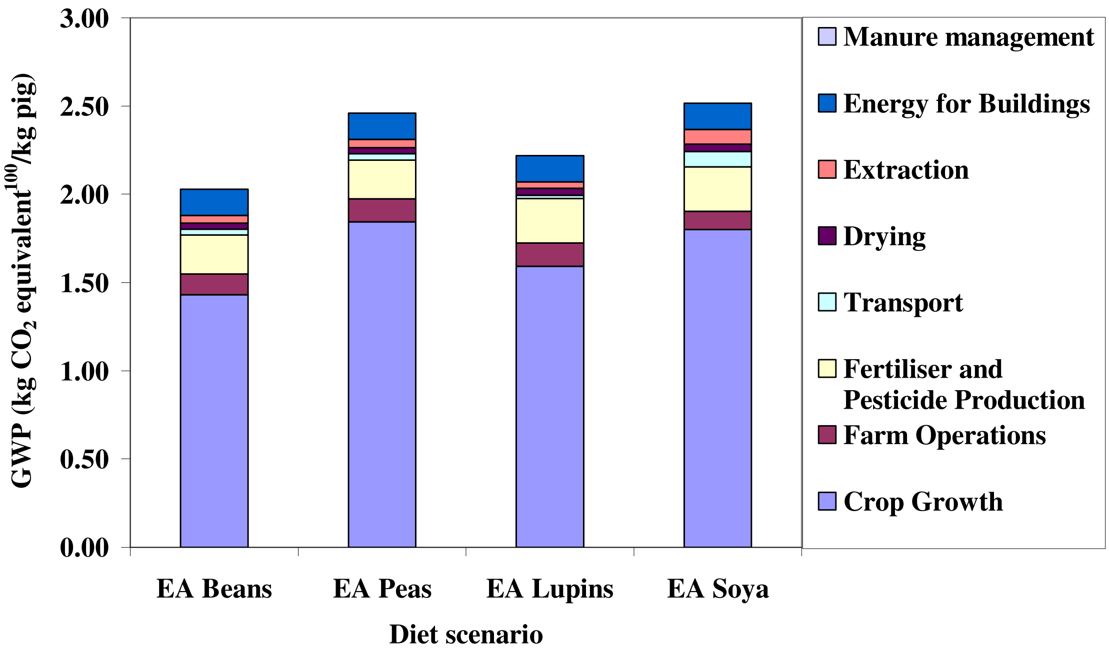

Figure 25 The contribution of all processes to the total GWP per kg pig at East Anglia in the synthetic fertilizer scenario.

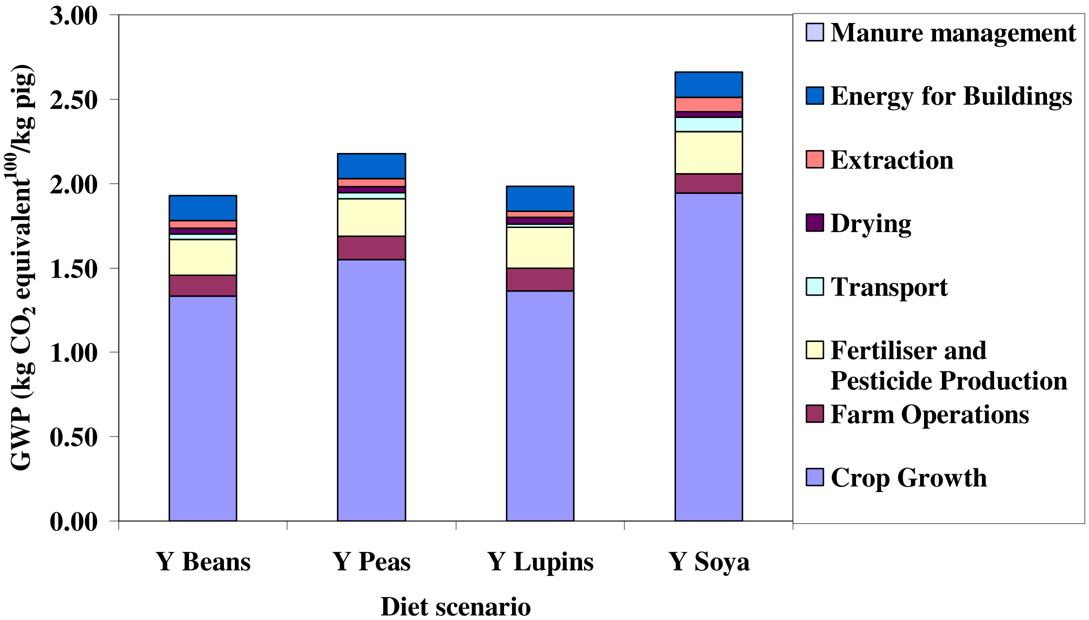

Figure 26 The contribution of all processes to the total GWP per kg pig at Yorkshire in the synthetic fertilizer scenario.

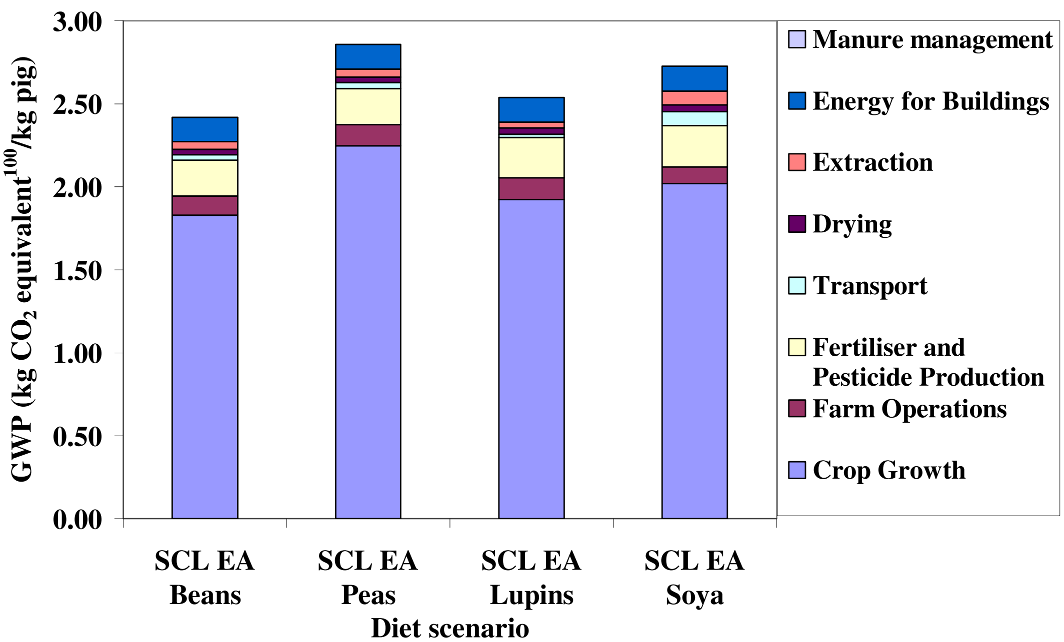

Figure 27 The contribution of all processes to the total GWP per kg pig at silty clay loam East Anglia in the synthetic fertilizer scenario..

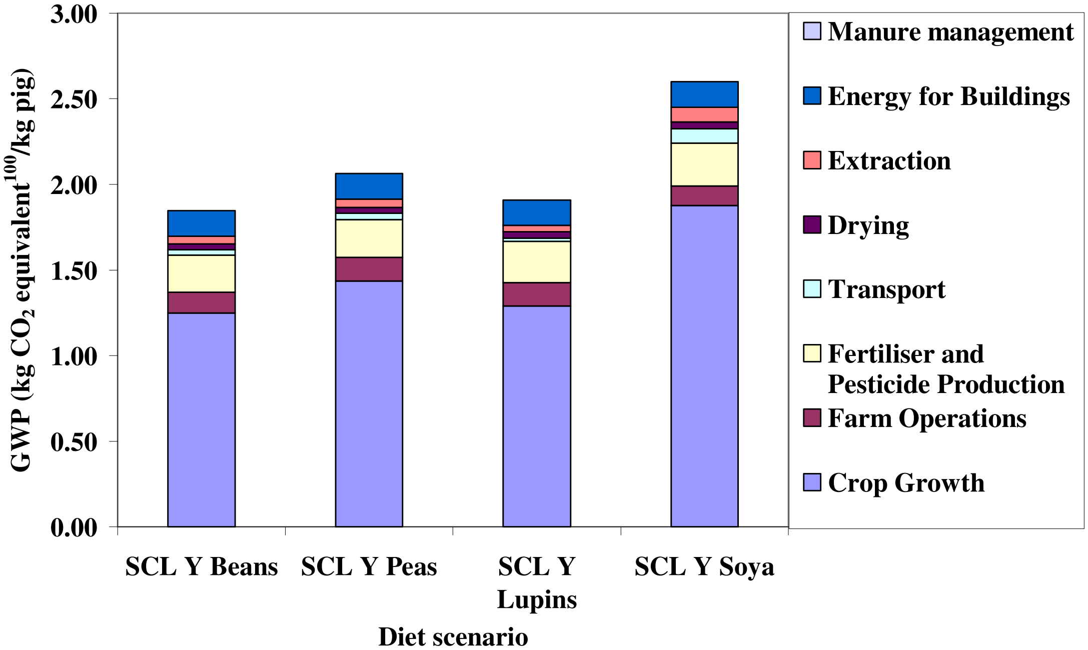

Figure 28 The contribution of all processes to the total GWP per kg pig at silty clay loam Yorkshire in the synthetic fertilizer scenario.

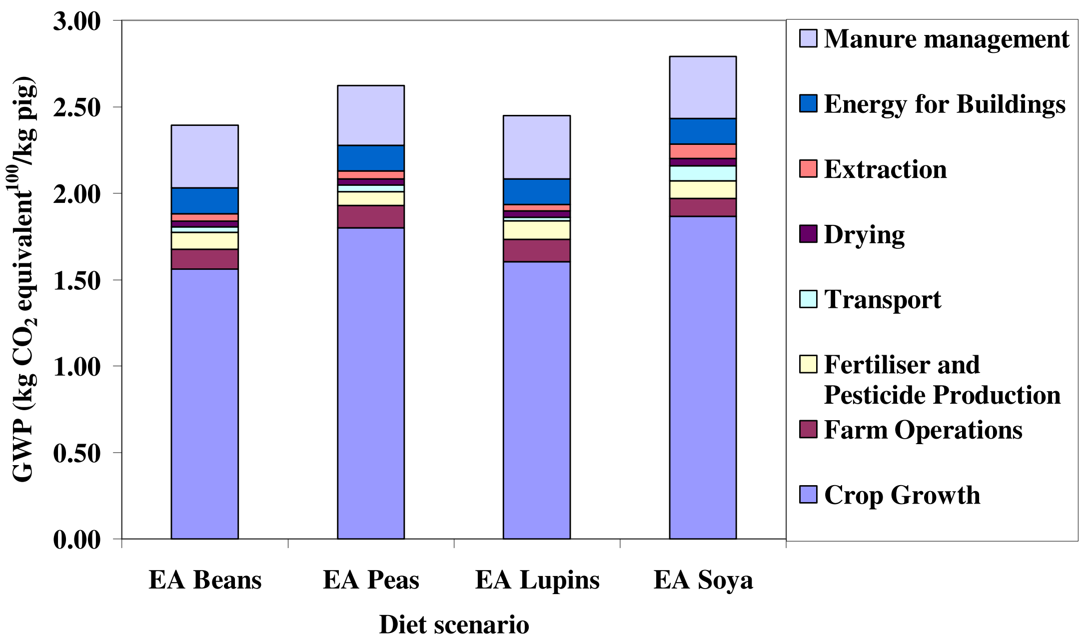

Figure 29 The contribution of all processes to the total GWP per kg pig at East Anglia in the slurry fertilizer scenario.

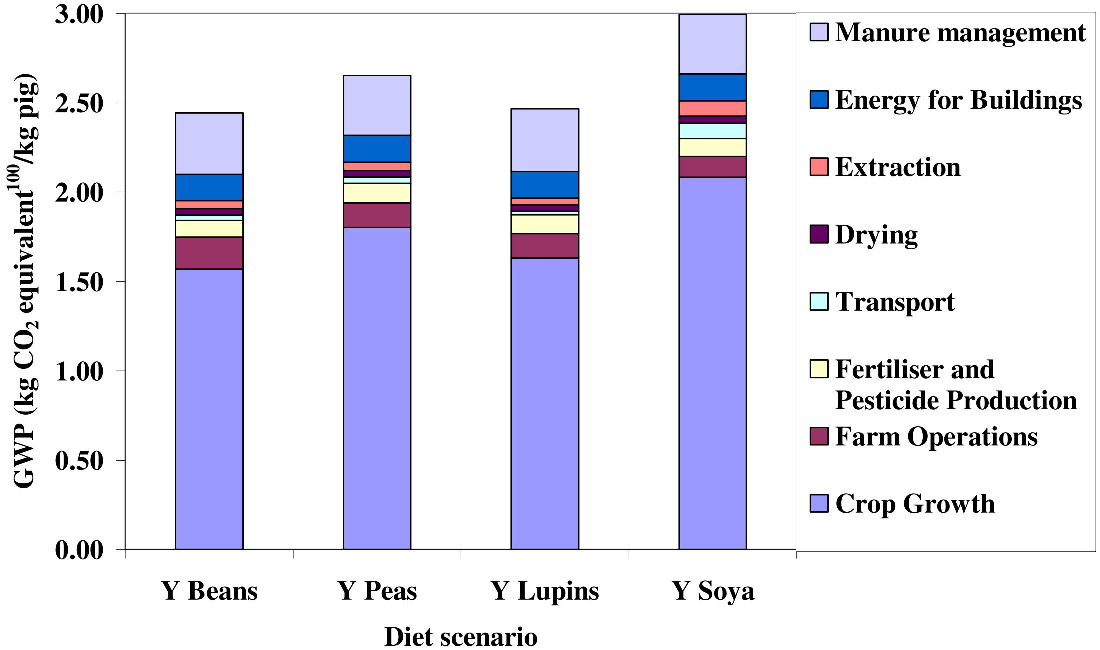

Figure 30 The contribution of all processes to the total GWP per kg pig at Yorkshire in the slurry fertilizer scenario.

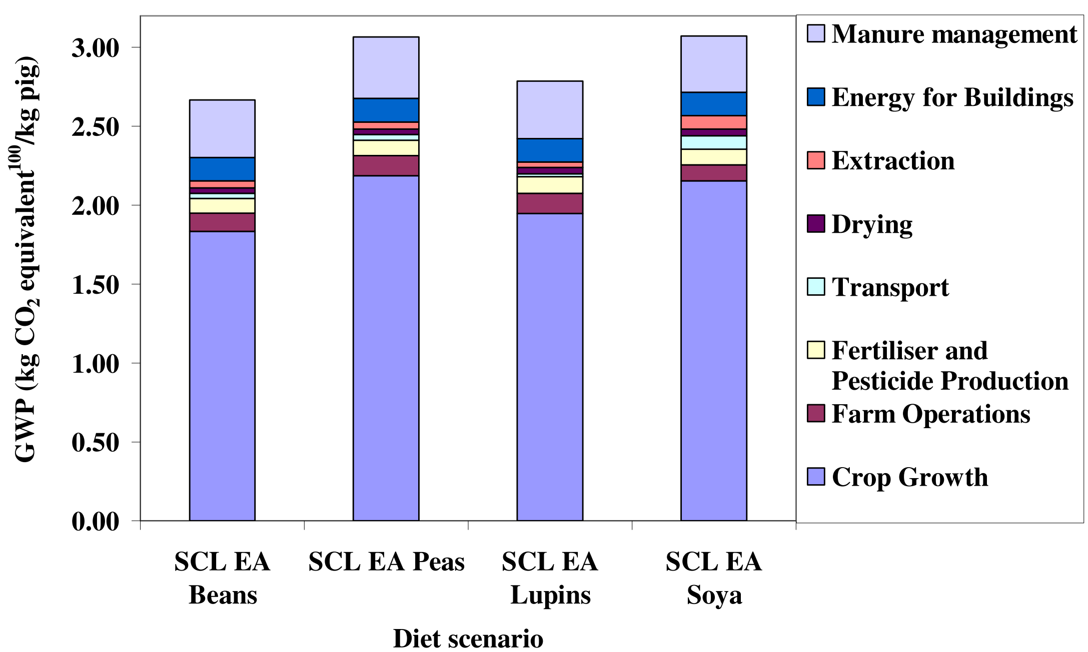

Figure 31 The contribution of all processes to the total GWP per kg pig at silty clay loam East Anglia in the slurry fertilizer scenario.

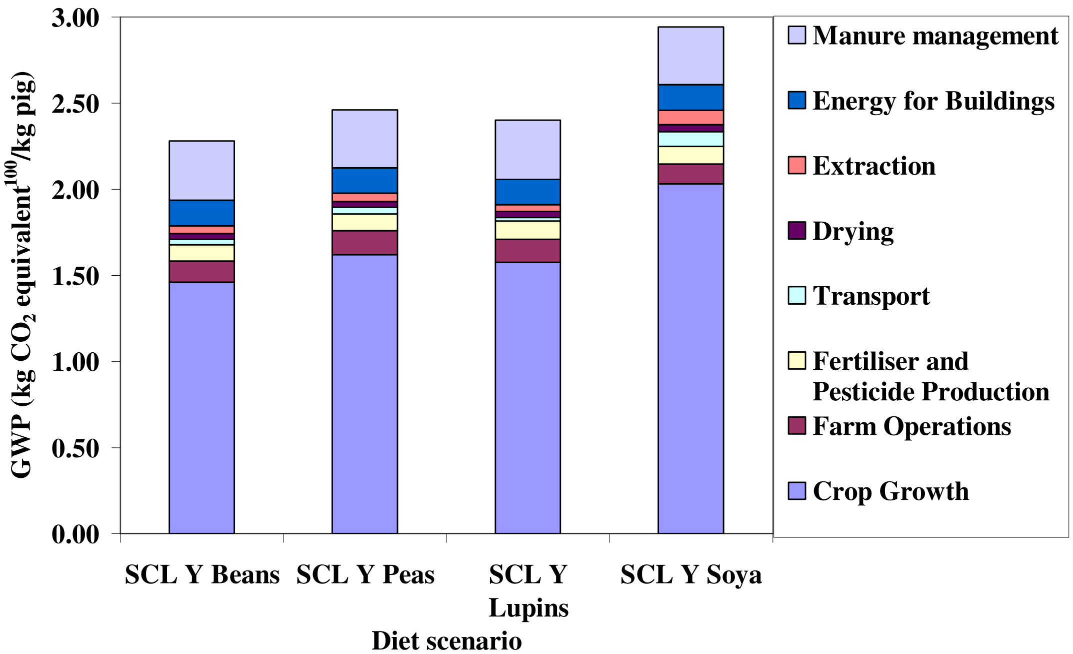

Figure 32 The contribution of all processes to the total GWP per kg pig at silty clay loam Yorkshire in the slurry fertilizer scenario.

The contribution to GWP from manure management is relatively high in the slurry fertilizer scenario, ranging from 0.334 to 0.389 kg CO_2 equivalent^{100}. In contrast, emissions from manure management were not included in the synthetic fertilizer scenario as this was outside the system boundaries. The energy requirements for housing is a constant value contributing to total GWP per kg pig in each scenario, this is 0.149 kg CO_2 equivalent^{100}. The environmental costs associated with the transport in the UK legume based diets are low, ranging from 0.02 to 0.04 CO_2 equivalent^{100} per kg crop (using the assumption of transporting crops 2km) whereas the soya crop transport from Brazil is included, 0.09 CO_2 equivalent^{100} per kg crop. The highest contribution from transport to total GWP occurred for the soya based diets, ranging from 3.1 – 3.4 % in the synthetic fertilizer scenario and 2.8 – 3.1 % in the slurry fertilizer scenario. Overall in the UK legume based diets, the transport contribution to the total GWP per kg pig ranged from 0.7 – 1.8 %.

As previously described, the highest contributor to total GWP per kg pig in both fertilizer scenarios is from the GHGs released during crop growth. The percentage proportions contributing to each result can be found in Appendix C.

3.2.2 Eutrophication Associated with Pig Production

Total eutrophication potential (kg PO_4 equivalent) associated with the production of 1 kg pig live weight is shown in Figure 33 to Figure 36 for each diet and site scenario for the synthetic fertilizer scenario and Figure 37 to Figure 40 for the slurry fertilizer scenario. In Figure 33 to Figure 40 DNDC outputs have been used to calculate the N loss during crop growth.

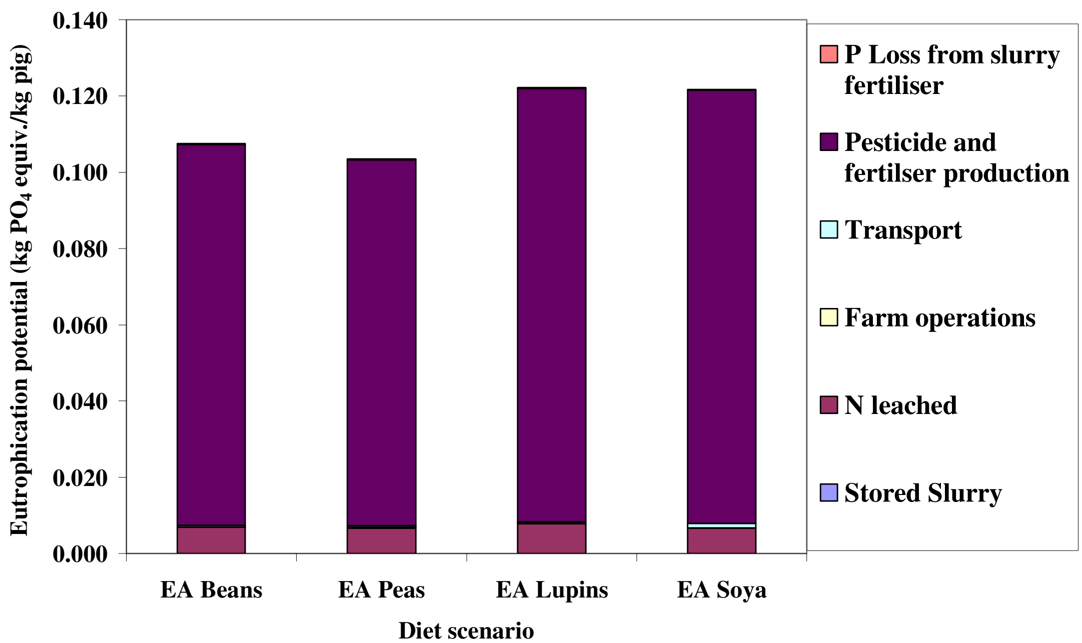

Figure 33 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in East Anglia in the synthetic fertilizer scenario.

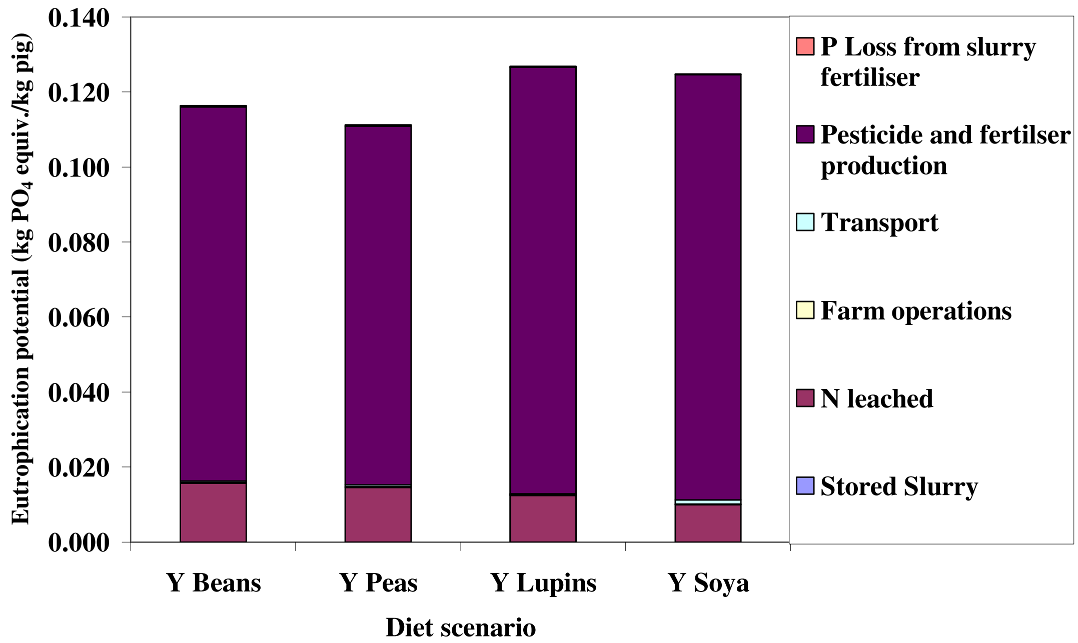

Figure 34 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in Yorkshire in the synthetic fertilizer scenario.

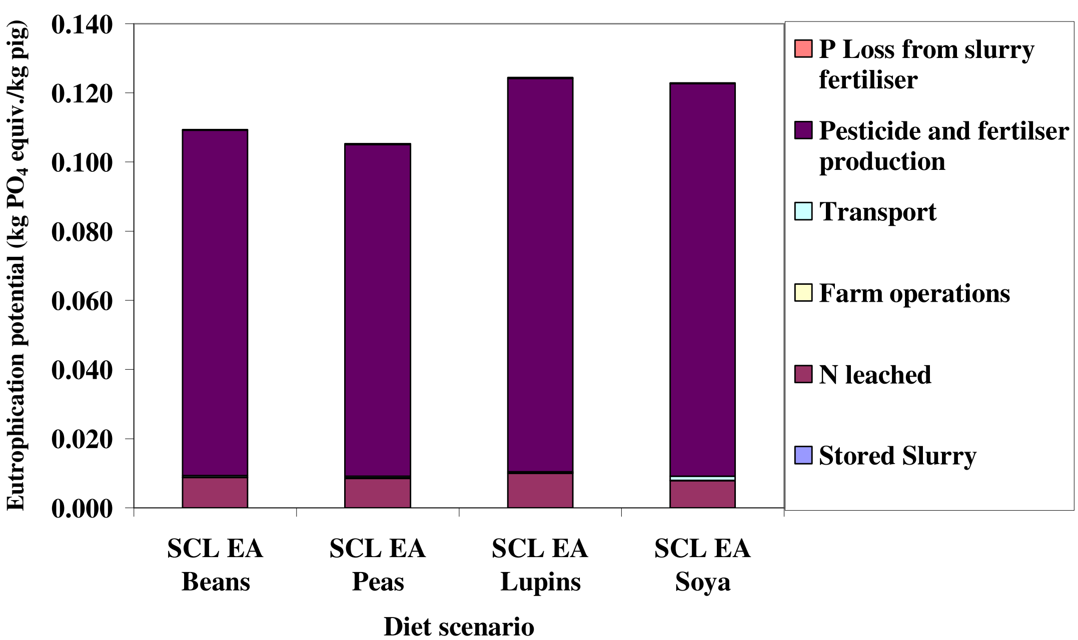

Figure 35 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in silty clay loam East Anglia in the synthetic fertilizer scenario.

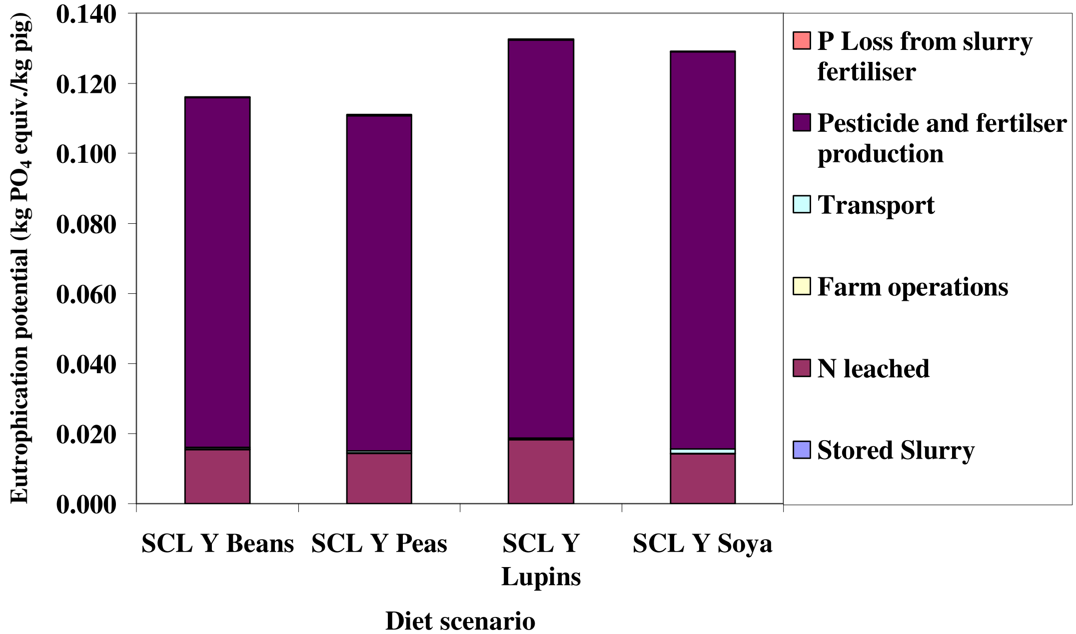

Figure 36 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in silty clay loam Yorkshire in the synthetic fertilizer scenario.

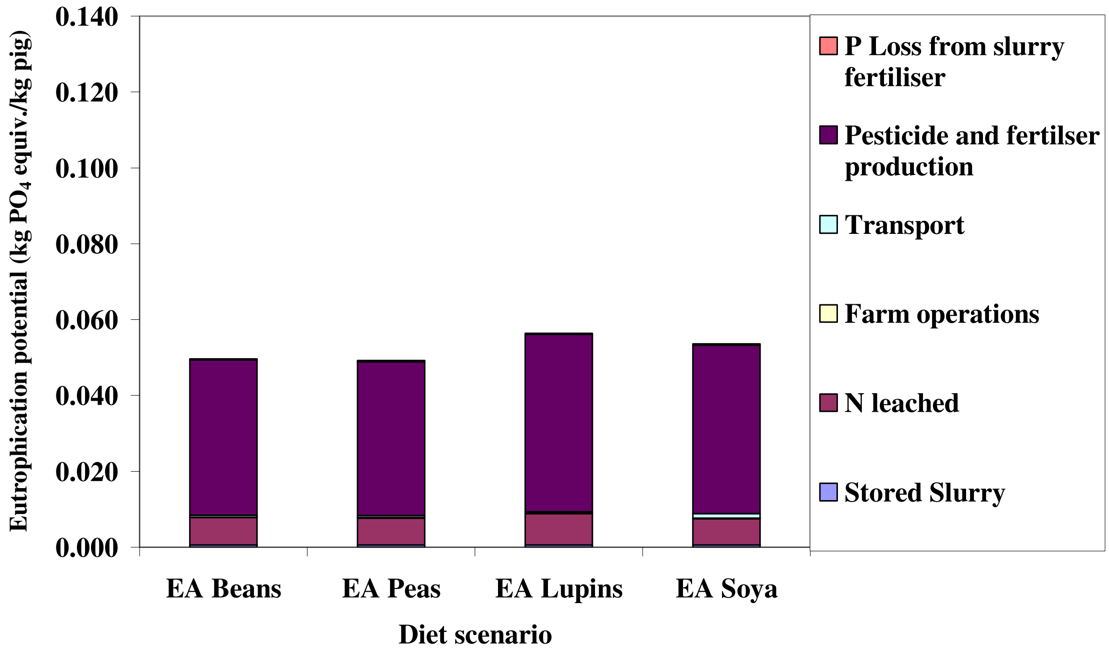

Figure 37 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in East Anglia in the slurry fertilizer scenario.

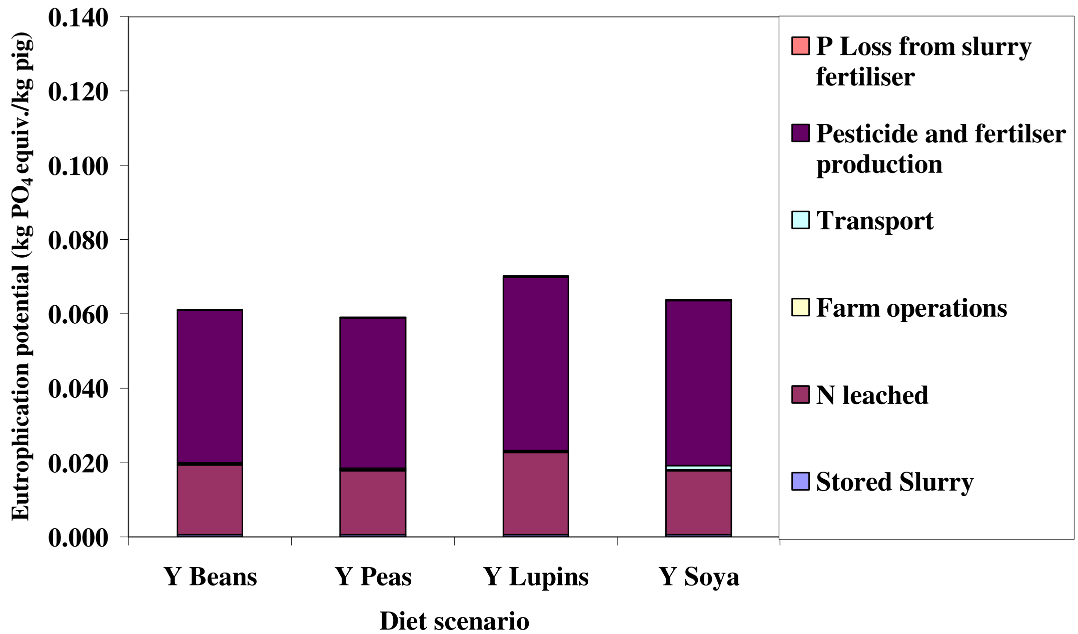

Figure 38 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in Yorkshire in the slurry fertilizer scenario.

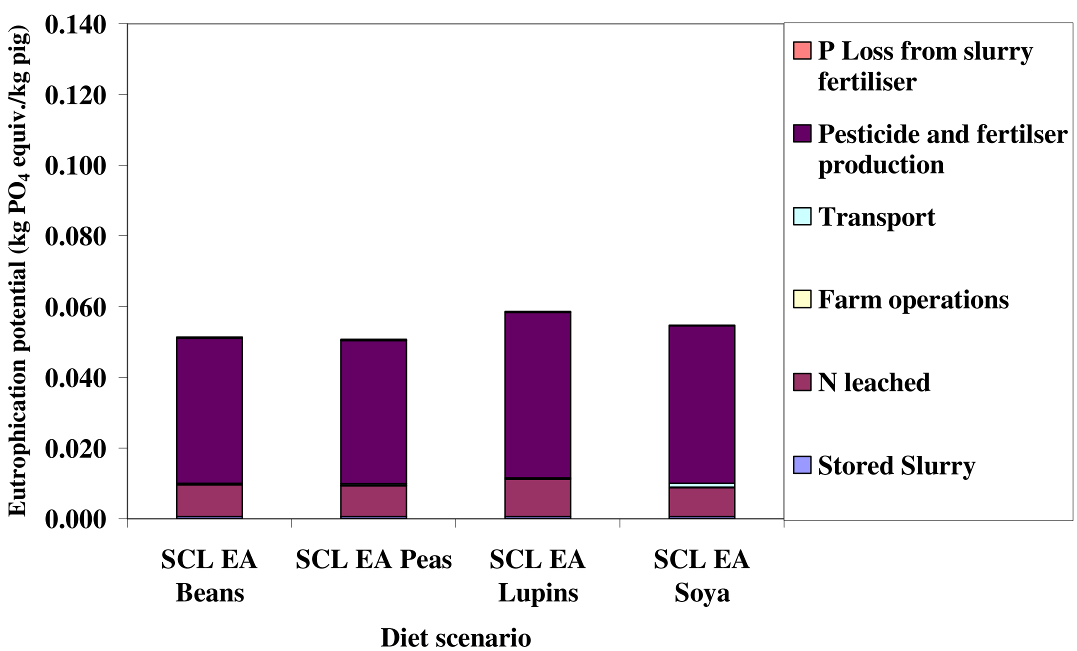

Figure 39 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in silty clay loam East Anglia in the slurry fertilizer scenario.

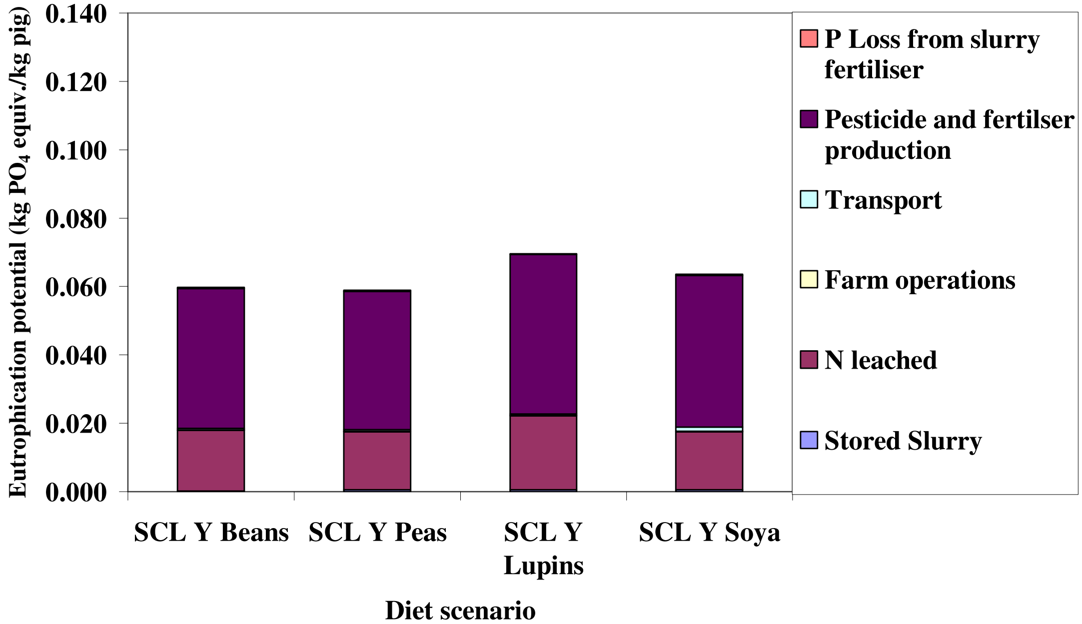

Figure 40 The contribution of all processes to the total eutrophication potential for the production of 1 kg pig in silty clay loam Yorkshire in the slurry fertilizer scenario.

The eutrophication potential (kg PO_4 equivalent) per kg pig live weight is considerably higher in the synthetic fertilizer scenario when compared with the slurry fertilizer scenario. The production of synthetic fertilizers contribute the most to total eutrophication potential, therefore in the slurry fertilizer scenario only 40 % of fertilizer requirements are supplied in synthetic form. This considerably reduces the total eutrophication potential per kg pig when compared with the synthetic fertilizer scenarios.

The Lupin based diets consistently have the highest eutrophication potential in both fertilizer scenarios, with the highest being 0.133 kg PO_4 equivalent at silty clay loam Yorkshire in the synthetic fertilizer scenario and 0.70 kg PO_4 equivalent at both Yorkshire and silty clay loam Yorkshire in the slurry fertilizer scenario. The pea based diets have the lowest eutrophication potential in both fertilizer scenarios, with East Anglia synthetic fertilizer scenario resulting in the lowest, 0.103 kg PO_4 equivalent and 0.049 kg PO_4 equivalent in the East Anglia slurry fertilizer scenario.

The highest contribution from transport to total eutrophication potential occurred from the soya based diet scenarios; 0.001 kg PO_4 equivalent. For the UK legume based diets, the contributions were small; 0.0003 kg PO_4 equivalent for the lupin based diets, 0.0004 kg PO_4 equivalent for the bean based diets and 0.0005 kg PO_4 equivalent for the pea based diets at all sites and fertilizer scenarios. Likewise with farm operations the contributions are small, ranging between 0.0001 to 0.0002 kg PO_4 equivalent at all sites and diet scenarios. N leached was calculated from DNDC outputs which varied between sites. The lowest amount of N leached occurred at East Anglia in both fertilizer scenarios, 0.007 kg PO_4 equivalent for all diet scenarios with the exception of the lupin based diet where the eutrophication potential of N leached was 0.008 kg PO_4 equivalent. The highest amount of N leached occurred at silty clay loam Yorkshire for both fertilizer scenarios, this ranged from 0.014 to 0.022 kg PO_4 equivalent. When comparisons are made between diet scenarios, the pea based diets consistently had the lowest amount of N leached and the lupin based diets the highest amount of N leached. In the synthetic fertilizer scenario, no account of slurry storage was included. Therefore the eutrophication potential from stored slurry was only included in the slurry fertilizer scenario, this was therefore consistent for each site and diet scenario, 0.0006 kg PO_4 equivalent. The percentage proportions contributing to eutrophication potential per kg pig at each site is given in Appendix D.

In Figure 33 to Figure 40 DNDC outputs have been used to calculate N losses from crop growth; however literature data suggest there is a 4 % loss of applied N. Therefore, in Figure 41 and Figure 42 a comparison has been made between N loss calculations; (1) N losses from DNDC and (2) using the assumption of 4 % N loss (Beusen *et al*. 2008).

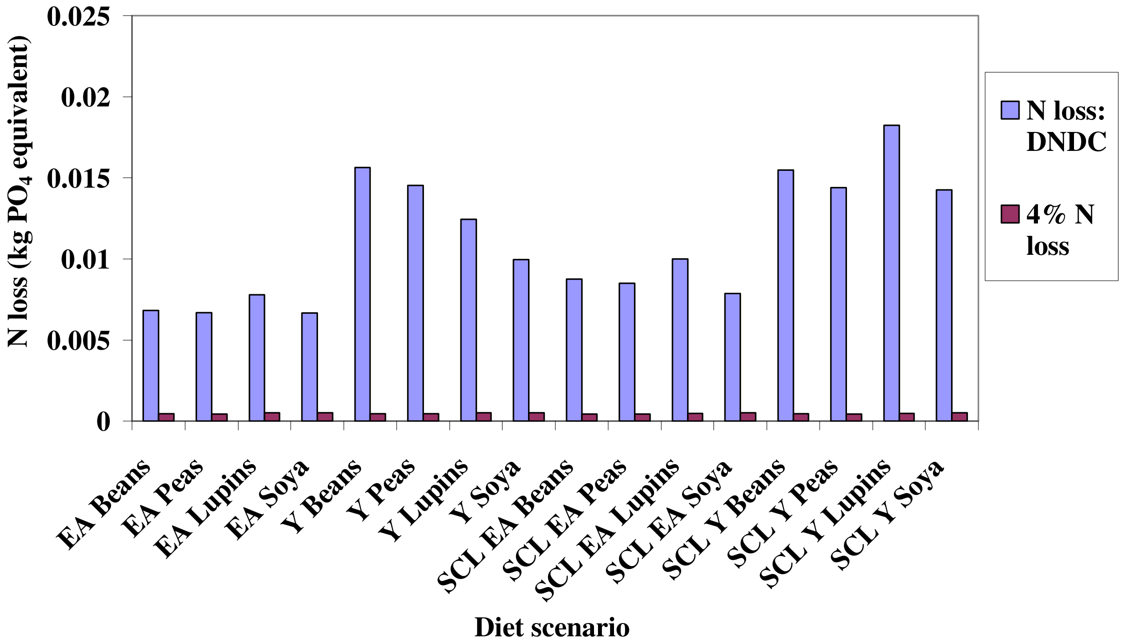

Figure 41 N losses from DNDC and the assumption of a 4 % N loss inputs in the synthetic fertilizer scenario.

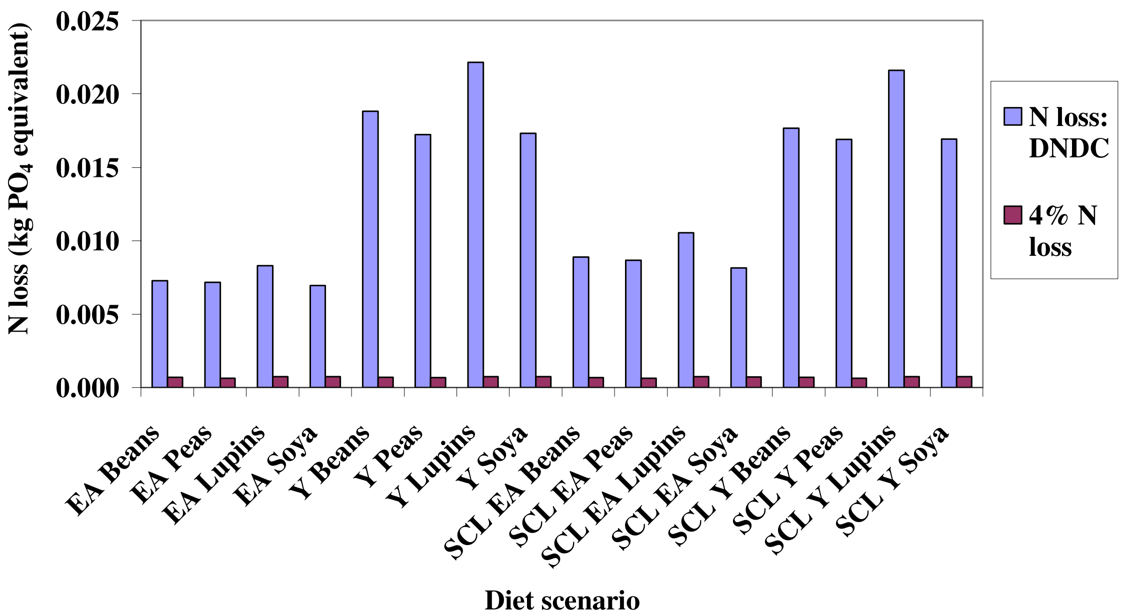

Figure 42 N losses from DNDC and the assumption of a 4 % N loss inputs in the slurry fertilizer scenario.

The N loss predictions from DNDC are distinctly higher than N losses using the assumption of 4 % N loss. The DNDC results show variations in N loss for each diet and site scenario, with a higher N loss in the slurry fertilizer scenario as it is based on total slurry applied. In general, there is higher N loss in Yorkshire compared with East Anglia, independent of soil type and therefore indicating this is a climate effect. N loss from the literature assumption remains constant throughout all site and fertilizer scenarios, showing that individual differences between site conditions are not considered.

3.2.3 Acidification Associated with Pig Production

The acidification potential associated with 1 kg of pig live weight in the synthetic fertilizer scenario is given in Figure 43 to Figure 46 and Figure 47 to Figure 50 for the slurry fertilizer scenario.

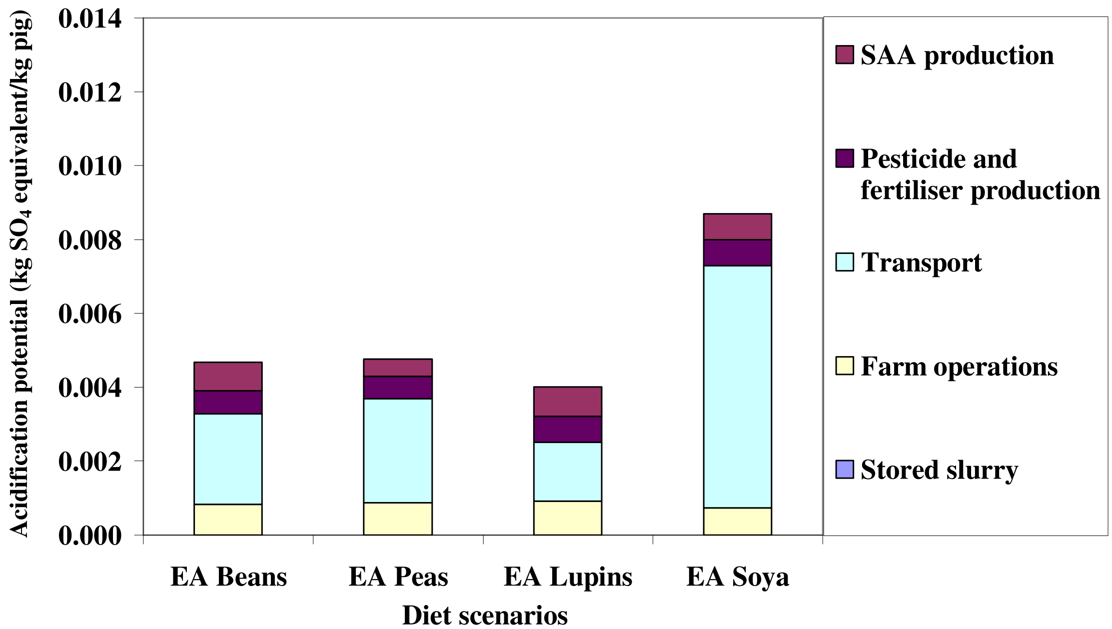

Figure 43 The contribution of all processes to total acidification potential in East Anglia in the synthetic fertilizer scenario.

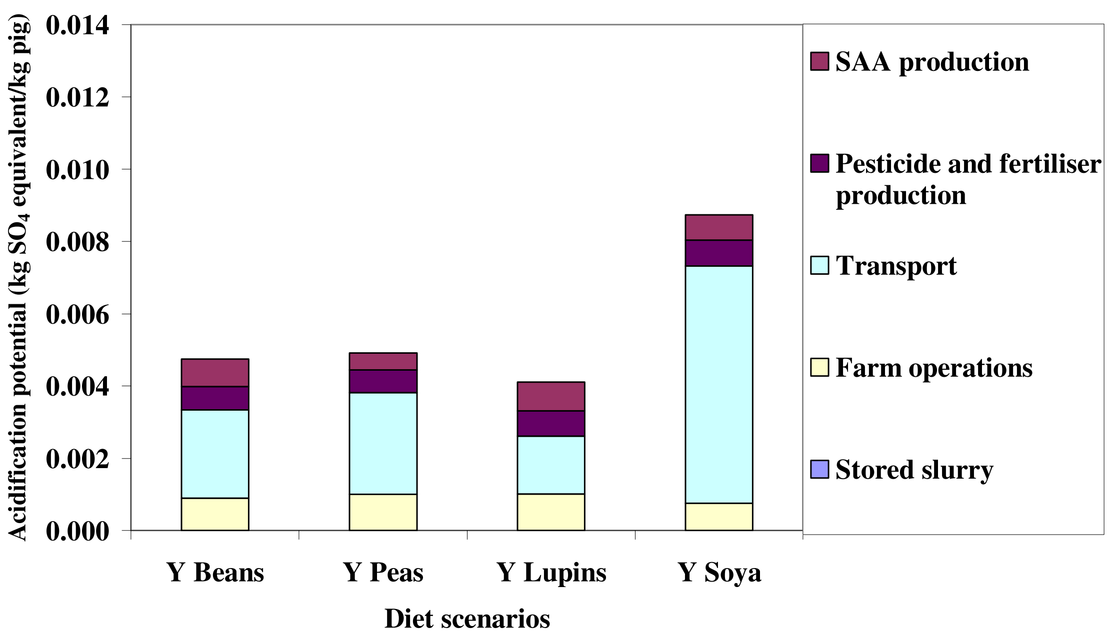

Figure 44 The contribution of all processes to total acidification potential in Yorkshire in the synthetic fertilizer scenario.

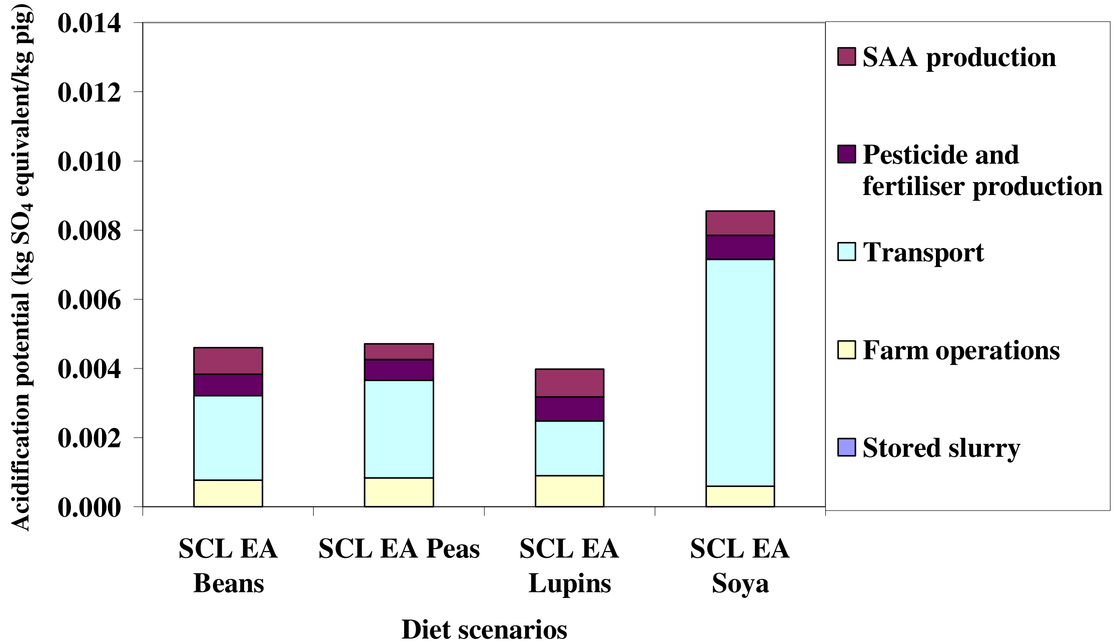

Figure 45 The contribution of all processes to total acidification potential for silty clay loam soil in East Anglia in the synthetic fertilizer scenario.

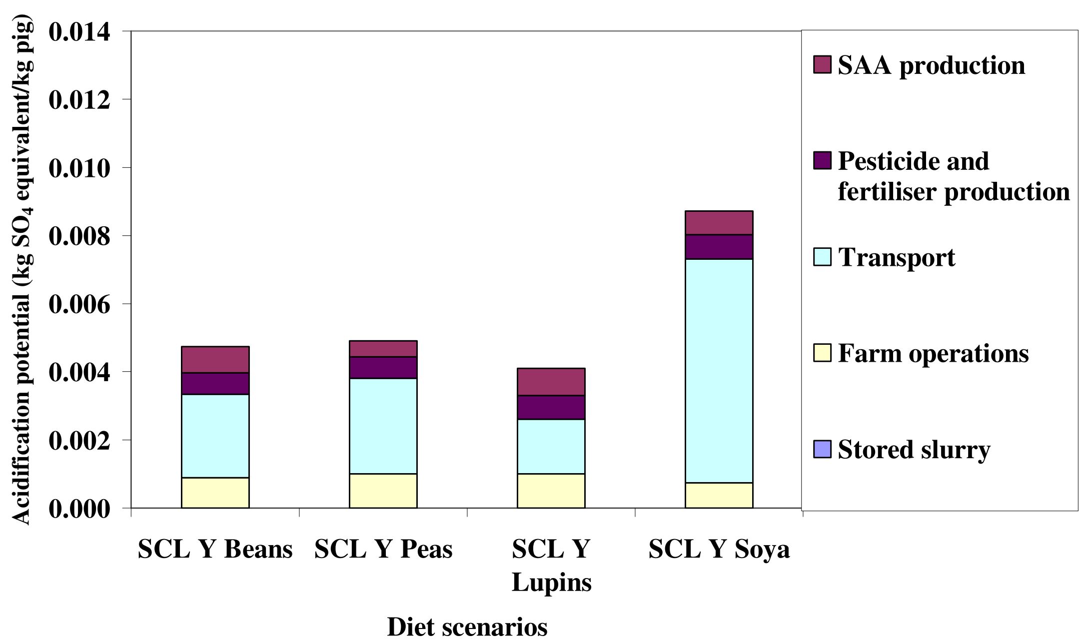

Figure 46 The contribution of all processes to total acidification potential for silty clay loam soil in Yorkshire in the synthetic fertilizer scenario.

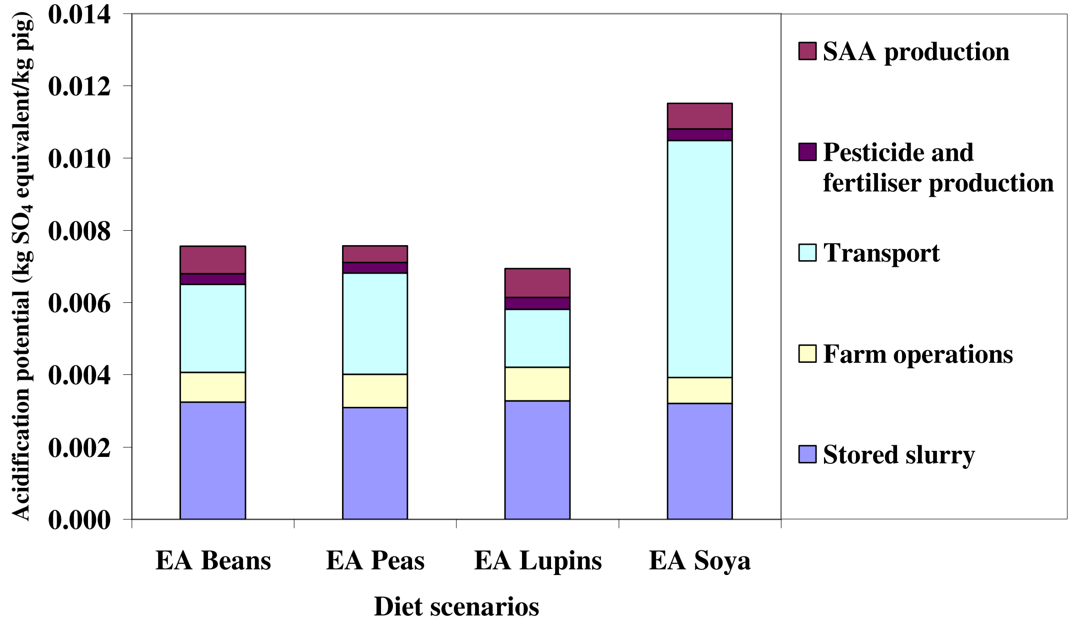

Figure 47 The contribution of all processes to total acidification potential in East Anglia in the slurry fertilizer scenario.

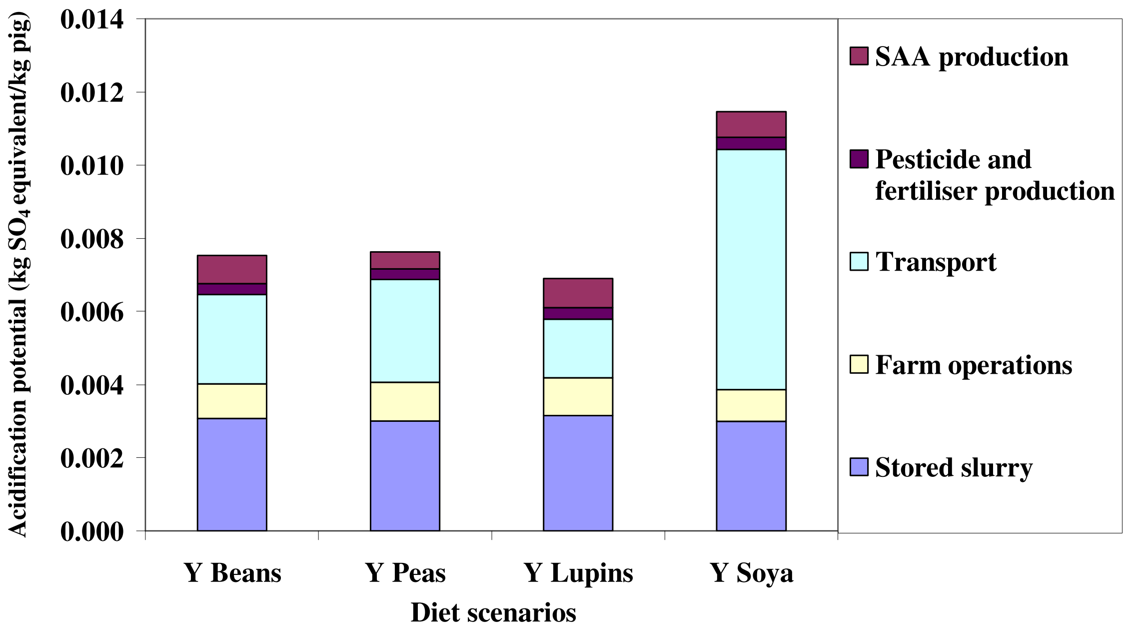

Figure 48 The contribution of all processes to total acidification potential in Yorkshire in the slurry fertilizer scenario.

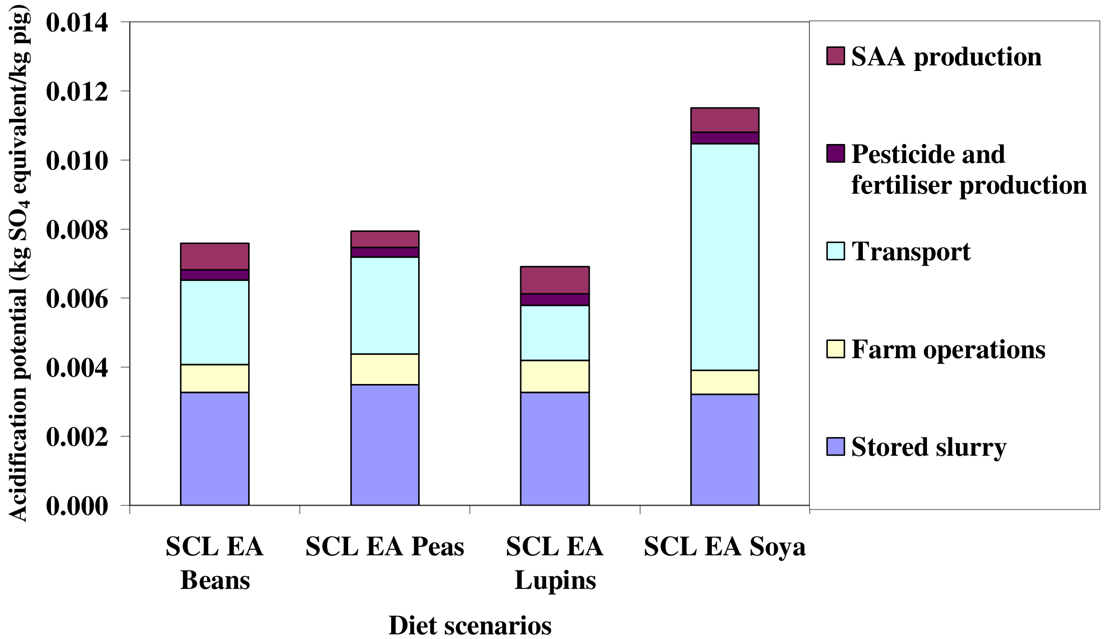

Figure 49 The contribution of all processes to total acidification potential for silty clay loam soil in East Anglia in the slurry fertilizer scenario.

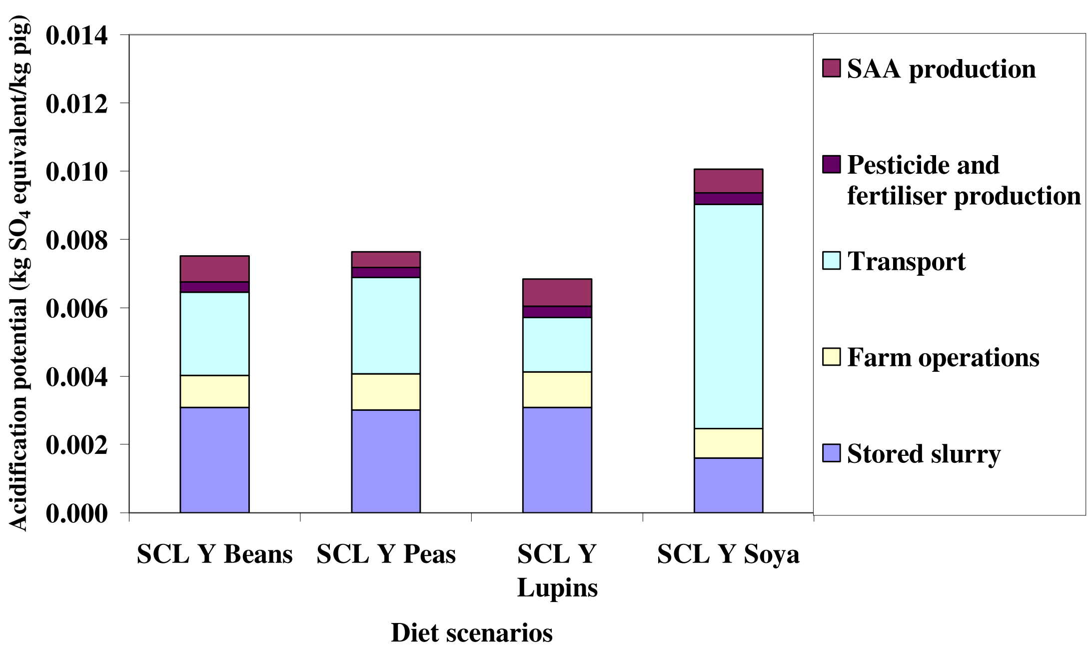

Figure 50 The contribution of all processes to total acidification potential for silty clay loam soil in Yorkshire in the slurry fertilizer scenario.

The variation between the total acidification potentials in both fertilizer scenarios for each diet and site scenarios are relatively small. In general, the synthetic fertilizer scenario systematically has the highest acidification potential for all scenarios. In both fertilizer scenarios, the soya based diets have the highest acidification potential per kg pig and the lupin based diets the lowest acidification potential per kg pig.

The contribution to total acidification potential from synthetic amino acid production was consistent for each diet scenario; 0.005 kg SO_4 equivalent in the pea based diets, 0.007 kg SO_4 equivalent in the soya based diets and 0.008 kg SO_4 equivalent in both the bean and lupin based diets. The contribution to total acidification potential from pesticide and fertilizer production was highest in the synthetic fertilizer scenario, ranging from 0.0006 to 0.0007 kg SO_4 equivalent. In contrast, in the slurry fertilizer scenario for all diet and sites the contribution was 0.0003 kg SO_4 equivalent. From transport, the contribution to total acidification potential is the highest for the soya based diet scenarios, 0.007 kg SO_4 equivalent. In comparison, for all home grown legume based diets, this ranges from 0.002 to 0.003 kg SO_4 equivalent. When comparisons are made between the contribution from farm operations at each site and diet scenario there is little variation, ranging from 0.0007 to 0.0011 kg SO_4 equivalent. As previously described, the contribution from stored slurry is only considered in the slurry fertilizer scenario, this is relatively constant across all diet and site scenarios, 0.0030 to 0.0035 kg SO_4 equivalent. The percentage proportions contributing to acidification potential per kg pig at each site is given in Appendix E.

The results presented show considerable variation between sites and fertilizer scenarios. The main cause of these variations is the direct result of individual site conditions, primarily due to crop production. Chapter IV will therefore describe in detail the reasons for these variations and highlight the reasons for modelling LCAs at farm level.

CHAPTER IV

4 Discussion

4.1 DNDC

From the DNDC simulations, yields were determined for crops at each site (using the average of all rotations which included the bean, pea and lupin rotations). The grain yields at each site were consistent with the average yields published in the SAC Farm Management Handbook (2009) for the UK. Small variations occurred between crop grain yields at each site due to differences between individual site conditions, primarily soil type and climate (Heinemann *et al*. 2002).

Each crop rotation was simulated in DNDC for twenty years by repeating each five year rotation four times. The Brazil corn-soya rotation was a two year rotation and this was repeated ten times. To determine the environmental impacts from DNDC outputs, calculations were taken from years fifteen to twenty of the rotation. The assumption was made that the soil composition in years fifteen to twenty were at a near-steady state, therefore C pools are consistent and no significant changes occurred in C pools after repetitions of the rotations (Fumoto *et al*. 2008). Daily weather data for each year of the UK rotations was used to simulate representative site conditions. Due to lack of available data, two years of daily weather data was used in the Brazilian rotation.

DNDC is a dynamic and deterministic model and can be regarded as an IPCC tier III approach for estimating N_2O emissions and changes in soil C. DNDC models the C and N bio-geochemistry in soils during crop growth and predictions of GHG emissions. DNDC was developed to model decomposition and denitrification processes which are influenced by the environment. Hence, the model can be used here to predict emissions from soils using site specific data for climate, soil and agricultural practices. DNDC specifically models denitrification (conversion of nitrate to nitrogen gas) and nitrification (microbial oxidation of ammonia) occurring in the soil (Li 2000). Using other more simplified calculations or models to determine GHG emissions from soils may not reliably represent what is actually occurring at specific sites due to the generalized assumptions made about site conditions.

Under aerobic conditions, bacteria decompose organic residues and microbial biomass which results in the production of soluble C and NH_3 (li 2007a and Li *et al.* 1992). Nitrification occurs under aerobic conditions, NH_4 can be oxidized to NO_2 and NO_3 by ammonium oxides and the potential rate of nitrification is related to the available NH_4, soil temperature and soil moisture. Denitrification occurs under anaerobic conditions and this occurs primarily in wet soils when bacteria utilize nitrate in the absence of oxygen. N_2O is primarily derived as an intermediate product of microbial denitrification and nitrification (Li *et al.* 1992). The composition of the soil is the driving factor for the amount of N_2O emissions. Within the DNDC calculations, the soil profile (Table 9) is used to calculate nitrate, nitrite, ammonium, organic residues, microbial biomass and also emissions of CO_2 and NH_3 (Li *et al.* 1992). DNDC is a globally recognised model for the predictions of N_2O emissions from agriculture soils (Li *et al.* 1992, Li *et al.* 2000, Lui *et al.* 2006, Li 2007a).

In contrast, the Canadian LCA used IPCC methodologies (adjusted to Canadian soil conditions and therefore equivalent to Tier II methodology) to calculate GHG emissions from soils. It is important to note that the Canadian LCA did asses two regions within Canada, but not to the level of detail as the present LCA. It is possible that the IPCC default methodologies may over or under estimate the emissions from soils due to the generalized assumptions. Hence the implementation of more detailed methodologies at sites (such as DNDC) maybe a useful addition when developing LCAs**.** Li *et al* (1992) observed from field experiments that emissions of N_2O are not continuous and that the release of N_2O increases following periods of rainfall and irrigation events. This occurs as a result of biochemical processes related to microbial activities in the soil (Li *et al.*

1992), when the pores in the top layer of the soil become saturated by water and the diffusion of atmospheric O_2 is blocked. The soil microbes absorb O_2 remaining in soil pores and hence deplete O_2 levels, which at the same time stimulates denitrifying microbes in the soil (li 2007a). These denitrifying microbes use NO_3 as electron acceptors (allowing the denitrification process to occur) and convert NO_3 into N_2O, NO and N_2 (Flechard *et al*. 2007; li 2007a). As previously described, using IPCC conversion factors does not account for site variations.

The GHGs emitted during crop growth are N_2O and CO_2. However, it should be noted that CH_4 is not produced in any of the rotations, as it is an end product of the fermentation of organic C under anaerobic conditions. Anaerobic microbes mediate the reduction of C to CH_4, a process that is optimized when temperatures range between 30 and 40 ºC which rarely occurs in the UK. As previously described, soil moisture is one of the driving factors that controls the rate of decomposition and denitrification, which indicates the importance of climate data for the assessment of environmental impacts associated with crop growth.

Between the modelled sites there is variation in the soil clay content. This affects the rate of decomposition of organic matter in the soil because clay has the ability to absorb C and prevents C from being decomposed. During decomposition, C is oxidized to CO_2 and the N present is converted to NH_3 and volatilized into the atmosphere (li 2007a).

The daily outputs from DNDC include the C and N released from soils. These daily outputs were used to calculate the total CO_2 and N_2O released during the growth of each crop, using the start and end point of plant to plant date of all crops in the rotation to allocate all GHG emissions to a crop. This was considered to be the most appropriate method as each day of the five year rotation was included. It is important to understand the impact of the chosen cut-off points to account for all farming practices throughout the crop rotation. This can be illustrated by explaining how the environmental impacts associated with winter rapeseed were calculated. In all rotations, the winter rapeseed is grown from 20th September in year 3 to 10th July in year 4, the succeeding crop is winter wheat which is planted on 2nd October year 4. Since the environmental impacts were calculated from the plant to plant dates, the impacts allocated to winter rapeseed were calculated from 20th September in year 3 until the planting of winter wheat on the 2nd October in year 4.

Ploughing (or tillage in general) alters the decomposition rate due to disturbances of organic matter in the soil, resulting in mixing of the soil and increasing oxygen availability (Li 2007a). This impacts on the total soil organic matter (SOM), which is important for the maintenance of soil fertility and plays a vital role in C sequestration (Liu *et al*. 2006). During DNDC simulations, the timing and type of tillage processes are entered and the effects of the resulting soil disturbance are included in the calculation of gas emissions. This means that the environmental impacts allocated to the growth of winter rapeseed is not restricted to the actual growing period but also includes the entire period to the planting of the succeeding crop. An alternative method could be used which divides the environmental impacts from the fallow period equally between the preceding crop and the succeeding crops.

Although DNDC is a useful model to calculate soil C and N dynamics, other nutrients such as P and K are not considered. Therefore, assumptions were made in the present study to determine the environmental impacts from P and K fertilizers. This inevitably results in discrepancies between the levels of detail in the methodologies used to model nutrient flows. Literature assumptions with regards to P and K inputs do not take into account site specific conditions whilst these are considered for C and N flows in DNDC.

This approach differs from the one used by Eriksson (2004), who used substance flow analysis methodology in the LCA framework to calculate the environmental impacts from crop production. This model included processes such as the energy required for farm processes and all nutrient inputs. However, soil processes were modelled using a simpler approach. For example, calculation of environmental effects associated with peas were modelled by assuming a positive fertilizing effect to the soil during crop growth due to the N fixing ability of peas, which in turn was accounted for by including a reduction in synthetic fertilizer applications to the succeeding crop. Similarly, the calculation of GHG emissions in the Canadian LCA was also relatively simple in comparison to the present study. Specific soils were not modelled *per se*, but instead IPCC conversion factors adjusted to Canadian conditions were used (equivalent to Tier II methodology). The GWP for peas in the present LCA was higher in the East Anglian sites 0.64 to 0.98 kg CO_2 equivalent^{100}, which does not correlate with the GWP for peas presented in the LCA for Swedish pig production, 0.31 kg CO_2 equivalent^{100} (Eriksson *et al*. 2004). However at the Yorkshire sites the GWP for peas is comparable with the results in the Swedish study, 0.20 to 0.46 kg CO_2 equivalent^{100}. The most plausible explanation for these differences is due to the variation in the methodologies used between the two studies. Within countrywide LCAs there is no indication of site specific conditions and therefore these LCAs do not reflect the variations which are likely to occur between soil types and climate. Additionally, the LCAs that have used IPCC conversion factors do not account for GHGs emitted from N fixed by legumes. DNDC does calculate GHGs emitted from N fixed by legumes which contributes to the higher estimates of GWP per kg crop in the present study.

To conclude the current findings with regards to the DNDC modelling, it has been highlighted that the predictions made by previous LCAs may only consider the average country conditions and not variation at specific sites within a country due to variations between soil types and climatic conditions. The present LCA has demonstrated that such regional variation within the UK exists and could be considered as a more representative model for calculating environmental impacts of crop production at specific sites.

4.2 Animal Growth Model

In the present study, the Animal Growth Model simulates the growth of the pig from conception onwards. However, within the system boundaries of this LCA only the period from 12 kg to 105 kg was used. This was decided as from weaning (approximately 7 kg) pigs are fed complex diets where ingredients such as fishmeal are used which are outside of this LCAs boundary. As this period lasts approximately 22 days it was assumed this would not have a large impact on the overall environmental impact per kg pig. Compared with the reviewed LCAs for pig production systems, only the Danish LCA used the same finishing weight while no two LCAs used the same starting weight. The Swedish LCA begins with pig weights of 29kg, the French begin at age 25.7 to 42 days (depending on the management system). The latter is a similar starting age as it was assumed that in the Animal Growth Model at 36 days of age a pig would weigh 12kg.

The present model predicts daily weight gains, daily feed intake (including energy and protein requirements) and N excretion. Predicted live weight gain per day increases with age. The daily weight gain averaged 520g during the starter period, 796g during the growing period and 898g during the finishing period. The feed intake per kg of weight gain increases up to slaughter weight (105 kg). These outputs correspond with expectations in the pig industry under well managed systems.

In the present study, enteric CH_4 was not calculated within the Animal Growth Model. Instead a constant value was incorporated using the IPCC default emission factor, 0.006 kg CH_4 per pig per day (IPCC 2006) (equivalent to the IPCC Tier I methodology). The Canadian LCA also included the default IPCC emission factor 1.5 kg CH_4 per pig/year, however this standard value was indexed to each swine category using the ratio of the category weight and the average weight. For future improvements in the level of detail included in the present LCA, this method could be implemented to enable an improved representation of enteric CH_4 losses. However, the estimated contribution of enteric CH_4

to total GWP is low in the present LCA (0.2 to 0.3 %) and therefore this is not a major contributor to total GWP per kg pig. The current assumption is therefore considered appropriate.

4.3 Diets

The aim of the present LCA was to determine the environmental effects of producing pigs using several home grown legume crops in the diets to meet protein requirements. These diets were based on home grown peas, beans and lupins, which were compared with a conventional soya based diet.

Within the pig industry, soybean meal is a conventional source of dietary protein due to its high crude protein (CP) content (44 %) and useful amino acid profile. The homegrown legumes do not match this high CP content or amino acid profile and instead have protein contents of 20.5 % (peas), 25.5 % (beans) and 34 % (lupins). Limitations apply to the inclusion levels of these legumes crops due to Anti-Nutritional Factors (ANF), these limitations were accounted for in the diet formulations. In all cases, these crops cannot meet the CP requirements of the pig alone due to the limited inclusion levels and additional soybean meal, rapeseed or SAA were included in to the diets. However, there may be scope in the future to develop new varieties of home grown protein crops with higher CP content, improved amino acid profiles and lower ANF content so that higher levels can be included in diets with the aim of replacing more soya in diets.

Diet composition influences the total environmental impacts per kg pig in each scenario. The main causal factor for this are the varying proportions of each feed ingredient included in diets. The French LCA (Basset-Mens & van der Werf 2005) led to similar conclusions. Certain feed ingredients have higher environmental impacts than others, therefore when feed ingredients with high environmental impacts are included at high levels in diets, this consequently increases the total environmental impacts for that diet scenario.

The primary consideration in the formulation of the diets was to ensure that the diets contained the same CP level as the conventional soya based diets. In addition, diets were formulated so that requirements for Digestible Energy (DE), total lysine, methionine, threonine and tryptophan were met and maximum fibre levels were not exceeded. The result of this meant that the proportions of cereal crops were not held constant across diet formulations. However, as this model was designed to correlate to practical conditions, this was considered to be the most appropriate approach. For example, the pea based diet contained the highest proportion of barley, the lupin based diet the highest proportion of wheat and the soya based diet contained the highest proportion of wheat feed. Large variations occurred between GWPs of feed ingredient which considerably impacts on the total GWP per kg pig in each diet scenario. For example the GWP of wheat ranges from 0.48 to 0.62 kg CO_2 equivalent^{100} and wheat feed ranges from 0.17 to 0.28 kg CO_2 equivalent^{100}. In all home grown protein based starter diets, soya was included to meet the CP requirement. The Swedish LCA assessed several diet scenarios, one of which assessed the environmental impacts of reducing dietary energy by 10 %. By doing so, N excretion was reduced by 15 % and acidification by 20 %. However, this assumed that pig performance remained the same, which is unlikely and this was not addressed in that LCA. A second scenario included a diet which omitted the use of soya completely (CP in the diet was supplied using peas and rapeseed meal and it was assumed daily gains were the same), which consequently reduced the GWP of producing 1 kg of pig. Although, the acidification and eutrophication potentials remained high, the use of a similar soya free diet would be an interesting scenario to include in this LCA. This could be attractive for UK pig producers to allow for a completely UK produced diet. However, giving the inclusion limitations of rapeseed meal and home grown legumes, other sources of protein would be required, specifically in starter diets or high inclusion levels of SAA would be needed. This however may increase diet costs depending on commercial SAA prices. In conclusion, although it may seem attractive to replace all soya in pig diets with home grown protein sources, failure to take account of the effects of such diets on pig performance can easily lead to overestimation of the environmental benefits.

4.4 Additional Processes

The environmental impacts of additional processes included in the pig production systems do not impact greatly on total GWP and eutrophication per kg pig. However they do impact considerably on total acidification due to the high fossil energy demands. These additional processes were included in the integration sub-model using IPCC Tier II equivalent methodologies. In the present study alternative energy sources such as wind was not considered. However in future developments of the model, this could be included to determine if renewable energy sources positively impact on the environmental impacts.

4.4.1 Manure Management

The manure management component of the system contributes significantly to total GWP per kg pig in the slurry fertilizer scenario. This includes the GWP associated with slurry storage and transport for exporting excess slurry. In the synthetic fertilizer scenario manure management was not included in the system boundaries. The GWP was calculated from the amount of slurry stored per year, therefore in the slurry fertilizer scenario the level of stored slurry fluctuated as slurry was applied to crops. The contribution to total GWP per kg pig therefore ranged from 0.334 to 0.389 kg CO_2 equivalent[100]. The primary GHG associated with slurry storage is CH_4. The average amount of CH_4 produced from stored slurry was 1.3 kg per pig (105 kg). Comparisons were made with results from the Danish LCA for slurry storage which concluded that from CH_4 emissions *per se*, 1.9 kg was produced per pig (100 kg) (Dalgaard *et al*. 2007).

In this LCA, it was assumed that the slurry was stored in a covered tank. The Danish LCA calculations were made using IPCC (2006) emission factors. Therefore this highlights that the IPCC guidelines do not consider the variations between slurry management systems. This could lead to misinterpretation of results, if for example the assumption was made that the slurry was stored in an uncovered tank, the GHG emissions would most likely be higher and IPCC emission factors do not consider such variations. The impact of manure management on total eutrophication potential per kg pig is low, ranging from 0.4 % to 1.3 % (0.001 kg PO_4 equivalent in all scenarios).

The environmental impacts occurring from slurry application to crops is calculated from DNDC simulations and fertilizer inputs. Therefore two separate calculations were used, this is calculated as N and P leached. In the slurry fertilizer scenario, N leached ranged from 0.007 to 0.22 kg PO_4 equivalent. The contribution to total acidification potential from manure management is high, ranging from 15.9 % to 47.4 % of the total acidification potential per kg pig or 0.003 kg SO_2 equivalent in each scenario. Emission factors are used to convert NH_3 to So_x equivalent, which is calculated from the amount of slurry stored per year whilst taking into account what is applied to crops. From the previously reviewed LCAs, assumptions were made that slurry produced was applied to crops grown for diet production and no comparisons were made of a cropping system completely based on synthetic fertilizer applications. In the present LCA, the environmental impacts of slurry in the synthetic fertilizer scenarios was not included as this fell outside the boundary of the LCA. If the environmental impacts from slurry were included in the synthetic fertilizer scenario, it would not allow for a true comparison to be made between the fertilizer scenarios. Although the application of slurry to crops would most likely occur in a practical situation, in the present LCA the aim was to show the environmental impacts from both fertilizer scenarios to determine if any potential environmental benefits can be achieved from the types of fertilizers used.

4.4.2 Production of Fertilizers and Pesticides

The production of both fertilizers and pesticides is an energy expensive process which is a significant contributor to the total GWP and eutrophication potential. To calculate the environmental impacts of fertilizer and pesticide production conversion factors were sourced from Williams *et al* (2006). As would be expected, due the high fossil energy requirements, the contribution to total GWP per kg pig was higher in the synthetic fertilizer scenario, this ranged from 0.212 to 0.252 kg CO_2 equivalent[100]. In contrast, in the slurry fertilizer scenario only 40 % of crop N requirements were supplied in synthetic form and therefore the contribution to GWP per kg pig ranged from 0.081 to 0.108 kg CO_2 equivalent. Additionally, the contribution to total eutrophication potential per kg pig from fertilizer and pesticide production is higher in the synthetic fertilizer scenario, ranging from 0.096 to 0.114 kg PO_4 equivalent. Compared with the slurry fertilizer scenario, the range is between 0.040 to 0.047 kg PO_4 equivalent, as less synthetic fertilizers are applied to crops in the slurry fertilizer scenario. The conversion factors for synthetic fertilizers include the environmental impacts for the complete production process, which includes extraction to energy in the Haber process for converting N_2 to NH_3 and the transport of minerals Williams *et al* (2006). Likewise, when considering acidification potential the contribution to total acidification potential in the synthetic fertilizer scenario ranges from 0.00061 to 0.00071 kg SO_2 equivalent. In the slurry fertilizer scenario this ranges from 0.00028 to 0.00032 kg SO_2 equivalent. The production of fertilizers and pesticides in both fertilizer scenarios contributed the least in all environmental impact categories in the bean and pea based diets. Whereas, the environmental impacts from fertilizer and pesticide production in both fertilizer scenarios was consistently higher in the lupin and soya based diets. This is a consequence of the feed ingredients used in the diet scenario.

4.4.3 Transport

The environmental impacts occurring from transport were included in the LCA and variations between diet scenarios occurred. The environmental impacts from transport associated with crops include; (1) transport to the processing plant, which was assumed to be 2 km for each crop and (2) the long distance transport associated with soya, a total of 10,830 km (including sea and road transport). All calculations are initially expressed per kg of crop. Therefore, the variation between diet formulations and the proportions of crops included influences the environmental impacts from transport per kg pig. This can be identified in diets where soya is included in high proportions, specifically the soya based diets. The lupin based diets required the least amount of additional soya to meet CP requirements when compared with the other home grown protein diets. From transport alone the contribution to total GWP per kg pig ranges between 0.02 kg CO_2 equivalent^{100} and 0.036 kg CO_2 equivalent^{100} in the both fertilizer scenarios for the home grown legume based diets. This is compared with the soya based diets, in the both fertilizer scenarios and all diet scenarios the contribution from transport was 0.086 kg CO_2 equivalent^{100}. However, when the percentage contribution to the total GWP per kg pig is considered, there are differences between fertilizer and diet scenarios. For the home grown protein based diets in the synthetic fertilizer scenario this ranges from 0.8 to 1.8 % and in the slurry fertilizer scenario from 0.7 to 1.5 %. When the diet formulation is considered, the pea based diets have higher inclusion levels of soya when compared with the bean and lupin based diet and therefore the contribution from transport to GWP per kg pig is always higher. Conversely the lupin based diets have the lowest amount of soya included and therefore, the contribution from transport to GWP per kg pig is always lower. In both fertilizer scenarios the soya based diets resulted with the highest contribution from transport, 2.8 to 3.4 %. The contributions from transport calculated for each diet and site scenario are somewhat higher than that calculated in the Danish LCA, which concludes the transport of soya contributes 1 % of total GWP per kg pig. This contribution of 1 % correlates with the contribution of transport in the home grown protein based diets, but not the soya based diets. The Danish LCA however did not consider the effects of diet variation, but only used a conventional soya based diet scenario (Dalgaard *et al*. 2007).

Transport in general had a relatively low impact on total GWP per kg pig, however the contribution in the soya diets were higher when compared with home grown diets. Referring back to our starting point, home grown protein sources fair better for the environment in terms of transport. This report highlights that in isolation this is true (transport for soybean meal is more environmentally costly than transport of home grown legumes) but when expressed as a proportion of total GWP per kg pig, the contribution is small.

The contribution of transport to eutrophication potential was also minor, ranging between 0.2 % and 0.5 %, in the synthetic fertilizer scenario for the home grown legume based diets and between 0.9 % for the soya based diets. In comparison, in the slurry fertilizer scenario, the contribution of transport to total eutrophication ranges between 0.4 % and 1.0 % for the home grown legume based diets and 1.8 to 2.2 % for the soya based diets. As slurry management is omitted from the synthetic fertilizer scenario, the transport associated with application is not included and therefore per kg pig the GWP contribution from transport is lower. The release of NO_x and SO_x from diesel combustion during transport is included in the calculations of acidification potentials. In the synthetic fertilizers scenario, the contributions from transport (for all diet and site scenarios) to total acidification ranged from 38.8 to 57.3 % for the home grown legume based diets and 75.2 to 76.7 % for the soya based diets. In the slurry fertilizer scenario this ranged from 23.0 % to 37.1 % for the home grown legume based diets and 57.0 to 65.3 % for the soya based diets. The acidification potential associated with transport in the soya based diets was consistently higher than that associated with the home grown legume diets.

4.4.4 Buildings and Machinery

The environmental impacts arising from buildings were included as an additional process in the integration model, the energy use is a constant value used for each diet and site scenario is 0.15 kg CO_2 equivalent[100] per kg pig. This includes the electricity required for lighting and ventilation in the pig unit.

Additionally, the decision was made not to include any environmental costs within the LCA for the manufacturing of machinery, buildings and routine veterinary procedures. It was concluded it would be difficult to quantify the exact environmental impacts associated with machinery manufacturing which would be highly variable. This is also agreed by PAS 2050 and Vink *et al* (2003) in which they describe the difficulty in quantifying these components into one figure. They also state that this value will be negligible and hence a constant value would have been used in each scenario. It would therefore not greatly affect the variation between scenarios for the total environmental costs of the different system.

4.5 Environmental Impacts of Pig Production

4.5.1 GWP per kg Pig

Overall, the soya based diets have concluded with the highest GWP per kg pig. The synthetic fertilizer scenarios have in general a lower GWP per kg pig when compared with the slurry fertilizer scenarios. The higher GWPs per kg pig in the slurry fertilizer scenarios is caused by higher environmental impact costs associated with slurry management. In the synthetic fertilizer scenarios, no environmental costs were included for manure management. In contrast, within the slurry fertilizer scenario more GHG emissions occur from crop growth, more specifically higher N_2O emissions. The average contribution from crop growth to total GWP per kg pig in all diet and site scenarios is between 63.9 to 70 % in the slurry fertilizer scenario. However, in comparison within the synthetic fertilizer scenario the average contribution from crop growth to total GWP per kg pig in all diet and site scenarios is between 68.5 to 78.5 %. This occurs as the environmental costs associated with manure are not included in this scenario, thus concludes with a higher contribution from crop growth. N_2O emissions released during crop growth in the slurry fertilizer scenario are higher. To meet the N requirement of crops the amount of slurry applied is calculated using the available N in slurry rather than total N. This is accounted for in DNDC by defining the total N and available N content in the slurry. The N which is not utilized by the crop remains in the soil (total N – available N) and due to nitrification and denitrification it is overtime converted into CO_2 and N_2O, and consequently released from the soil resulting in higher GHG emissions. Thus more N_2O emissions occur in the slurry fertilizer scenario than the synthetic fertilizer scenario.

From the considered scenarios, there are clear differences between results of GWP per kg pig. DNDC outputs are the driving factors for these differences. More specifically the variation between CO_2 and N_2O released from soils during crop growth. The variation in GWP for feed ingredients between scenarios is driven by (1) the use of synthetic fertilizers or slurry (2) differences in soil properties, (3) variation in climate and (4) the exact timing of tillage and fertilizer applications in relation to climatic variables. The GWP of winter barley (per kg feed ingredient) was highest at Yorkshire. This was caused by heavy rainfall events during the growing season which negatively affected the yield of winter barley. To confirm that the high GWP of winter barley was due to the high rainfall prior to harvest, the weather file was altered, reducing the precipitation at days 223 (5.74 cm) and 224 (2.7 cm) to 0 cm. DNDC was run with this adjusted weather file and the N_2O emissions reduced from 5.5 kg N_2O/ha to 2.2 N_2O/ha (Appendix F). This demonstrated the effect of rainfall on estimated N_2O emissions. It maybe assumed that previous LCAs do no account for such variations and instead assume constant average conditions. However, in this LCA by modelling specific sites within the UK, identification can be made of areas within the UK which potentially result with higher

GWP from crop production, primarily caused by the combination of soil type and climate.

The previous LCAs which have used IPCC emission factors to calculate GWP do not account for N_2O emissions occurring from N fixed by legume crops. The IPCC calculations relate to the direct and indirect N_2O emissions associated with fertilizer applications. Therefore, as legume crops do not require N fertilizers there is subsequently no account for N_2O emissions from legume crops. However, biologically fixed N is still nitrified and denitrified over time and hence N_2O emissions do occur (Zhong *et al*. 2009). N_2O emissions from N fixed by legume crops are included within DNDC simulations and therefore explain why the GWP per kg legumes are higher in this LCA when compared with the results from the Swedish LCA for example.

The Swedish LCA concluded that peas have a GWP of 0.31 kg CO_2 equivalent^{100}, the results for legume crops in this LCA ranged from 0.20 to 1.00 kg CO_2 equivalent^{100}. Generally in the slurry fertilizer scenario, the legume crops have higher GWPs per kg feed ingredient when compared with the synthetic fertilizer scenario. A possible reason for this could be due to residues of N from slurry remaining in the soil for longer periods of time, which is less likely to occur when synthetic N fertilizers are applied. Legume crops may potentially emit N_2O from fixed N, but also biological N from slurry residues, consequently increasing the GWP per kg feed ingredient. There are also site differences; between the two modelled UK sites, the GWP of the legume crops grown in the East Anglian sites are consistently higher than the Yorkshire sites. This occurs due to variations in site conditions, soil type and climate. This is evident from DNDC outputs that the levels of both CO_2 and N_2O emissions in East Anglian sites. The silty clay loam soil type has been used as a common soil type at both Yorkshire and East Anglia, and at the East Anglian site the legume crops still have the higher GWP per kg feed ingredient. Therefore the differences are then a result of the climate variations, more specifically high rainfall during the year that the legume crops were grown. Studies by Flechard *et al*

(2007) and Li (2007) have shown that during periods of high rainfall increased amounts of N_2O emissions occur, agreeing with the results in this LCA.

When comparisons are made between the soya based diets and home grown legume based diets in both fertilizer scenarios, the soya based diets consistently have the highest GWP per kg pig and the bean based diets consistently have the lowest GWP per kg pig. When considering the home grown legume based diets *per se*, the GWP per kg pig is highest for the pea based diets. As previously described, in the slurry fertilizer scenario the GWP per kg pig is higher and between sites the results are comparable. However when both fertilizer scenarios are considered, the GWP per kg pig is the highest at silty clay loam East Anglia site. This identifies that the combination of soil type and climate can affect the overall GWP, more specifically from feed production.

Due to variations in diet formulations, feed ingredients that have a high GWP considerably impact on the overall GWP per kg pig. For example, as soya is used as the primary protein source in the soya based diets the high GWP per kg soybean meal (2.35 kg CO_2 equivalent^{100}) considerably increases the total GWP per kg pig. In the UK home grown diets, the pea based diets conclude a consistently higher GWP per kg pig. Again, this is dependent on the proportion of feed ingredients included in to the diet formulation. Peas are included into the diet at 30 % in the starter, grower and finisher diets. The pea based diets also include high levels of barley, which has the highest GWP out of all cereal feed ingredients in all scenarios, therefore this specific diet formulation impacts considerably on the total GWP per kg pig.

Wheatfeed concluded with the lowest GWP per kg feed ingredient in all scenarios. Therefore, diets formulated with high levels wheatfeed are associated with lower GWP per kg pig. For example, the bean based diets are generally associated with a lower GWP per kg pig in all scenarios due to the high inclusion of wheatfeed when compared with the other diets. In all scenarios the GWP per kg wheatfeed ranged from 0.17 kg CO_2 to 0.28 kg CO_2 equivalent^{100}. The GWP per kg wheatfeed was calculated based on economic allocation as a co-product from the wheat crop. Consequently, when diets are formulated with high (or maximum inclusion levels) of wheatfeed the contribution to total GWP per kg pig is overall much lower. The economic allocation method of crop co-products used in this LCA was seen as the most appropriate approach as economics is seen as the driving factor for commercial diet formulations. This economic allocation method was used in all of the previously described LCAs with the exception of the Canadian LCA which used a mass allocation approach.

The bean based diets are generally associated with the lowest GWP per kg pig. Beans are included into a diet at a lower level, compared with lupins for example, due to a maximum inclusion level of 20 %. However, the bean based diets include the highest proportion of wheat when compared with all other diets. Wheat also has a lower GWP ranging from 0.48 to 0.62 kg CO_2 equivalent[100] and consequently contributes to an overall lower GWP per kg pig in the bean based diet scenario.

The soya based diets in most scenarios concluded with the highest GWP per kg pig. The driving factor being the high inclusions of soybean meal which additionally has the highest GWP when compared with all feed ingredients. However, it must be highlighted that the GWP associated with soybean meal is independent of UK feed ingredients. Soya is a low yielding crop, therefore this consequently impacts on the overall GWP per kg crop (as the GHGs are divided by the yield/ha). When comparisons are made with the GWP of soybean meal and the results from the previously described LCAs, the GWP per kg soybean meal is higher in this LCA. The assumption was made that prior to soya growth, the land was previously used for arable production. It is important to be aware that this may not be the case. If, for example, the assumption was made that the land was previously forest land, the GWP per kg soya could potentially be even higher as the GHG emissions associated with land use change would be included. PAS 2050 assumes GHG emissions from land use change are released in equal annual amounts for 20 years. This would therefore increase the GWP of the soya crop further if this approach was adopted in this study. For future developments of the model a scenario which includes land use change, i.e. deforestation prior to soya production, could be considered.

With the aim of calculating the environmental impacts per kg pig as accurately as possible, the most appropriate data was sourced. For UK sites this was possible, however for Brazil, data was not as widely available and this leads to limitations in the calculations for soya production. Data for soil properties was sourced through the best possible means (Prof. Da Silva, 2009), however actual weather data for Brazil for years 1998 to 2007 could not be found. Therefore, instead of using ten years of actual weather data (repeated twice) one year of daily weather data was used (repeated 20 times). Williams *et al* (2006) used a similar approach in the UK LCA, a simulation model was used for crop growth using nine combinations of soil types and rainfall, however no specific detail was given on temperatures.

The total GWP per kg pig in the present LCA falls comfortably within the range of results from the previous LCAs for pig production systems described in Chapter I, ranging between 1.92 to 3.08 kg CO_2 equivalent^{100} per kg pig. In the reviewed LCAs the results range from 1.31 to 6.40 CO_2 equivalent^{100} per kg pig. Differences occur between the results of the described LCAs, which are caused by differing management scenarios modelled, LCA methodologies, assumptions and data used. Consequently this makes direct comparisons between LCA results difficult. There may be scope to reduce the GWP per kg pig in this study which may include formulating diets and considering the GWPs of the feed ingredients used in the diets.

4.5.2 Eutrophication Associated with Pig Production

The main contributors effecting total eutrophication potential per 1 kg of pig are (1) fertilizer scenario (2) soil characteristics and (3) climate associated with each site. For all scenarios, the eutrophication potentials were higher when synthetic fertilizers were used. The eutrophication potentials ranged from 0.049 kg PO_4 equivalent in East Anglia for the pea based diet in the slurry fertilizer scenario to 0.133 kg PO_4 equivalent for the lupin based diet at silty clay loam Yorkshire in the synthetic fertilizers scenario. Overall, the synthetic fertilizer scenarios concluded with the highest eutrophication potentials per kg pig, primarily caused by the eutrophication impact associated with the manufacturing of synthetic fertilizers. Therefore in the slurry fertilizer scenarios, the impact is lower as only 40 % of fertilizer requirements are supplied in synthetic form. Conversion factors were sourced from Williams *et al* (2006) and include (Table 5) the impacts associated with the production, packing and delivering of fertilizers. These conversion factors were used for each fertilizer type and calculated in relation to the amount of fertilizers applied to crops and then further calculations were made to determine the fertilizer requirement per kg feed ingredient. Therefore the total eutrophication impact ranged from 86.7 to 93.3 % in the synthetic fertilizer scenario. In the slurry fertilizer scenario the contribution to total eutrophication potential per kg pig is lower ranging from 66.5 to 83.1 %.

As previously described, the diet composition is the main cause of variation between scenarios. The lupin based diet included the highest proportion of wheat. Wheat has a high requirement for P fertilizer compared to other feed crops, and therefore there is a greater propensity for P leaching to occur. It has been assumed that 1 % of P fertilizer applied is lost through leaching (Chen *et al*. 2006). In the synthetic fertilizer scenarios, the P requirements per crop can be met exactly as fertilizer inputs directly meet requirements. However in the slurry fertilizer scenario this is not the case. Slurry is primarily applied to meet the N requirement of the crop. This therefore means that the P and K supplied by slurry are higher than the requirement per crop. Therefore as P and K have an increased risk of leaching, there is consequently a higher eutrophication potential associated with crop growth.

The higher N requirement of wheat also contributes to the increased eutrophication potential per kg pig due to higher amounts of leached N when compared with other crops. Additionally when high levels of wheat are included in diets, for example the lupin based diets. The N leached associated with wheat is then allocated to the diet scenario which subsequently increases the total eutrophication potential of the diet scenario. The N leached during crop growth had a higher contribution on total eutrophication potential in the slurry fertilizer scenario compared with the synthetic fertilizer scenario, this ranged between 13.0 % to 31.6 % and 5.5 % to 13.5 % respectively. While crops can only utilize the available N (NO_3) in the slurry, the N which is not required by the plant is potentially leached from the soil or added to soil pools. This is not the case when synthetic N fertilizer is applied, as almost all of N in synthetic fertilizer is available to the crop and is applied when crops can make best use of resources which consequently reduces the risk of N leaching after application (assuming suitable site conditions).

The N losses occurring from leaching were calculated from DNDC outputs. However assumptions can be applied to the proportion of N loss (using the available N input). A comparison was made between the N losses from DNDC outputs with an assumption of N loss taken from Beusen *et al* (2008) assuming a 4 % N loss from total N input into any cropping system. A large variation occurred between the results of the two modelling methods. N loss from cropping systems is dependent on individual site conditions. Therefore, using a fixed assumption of N loss does not reflect the individual site conditions. The calculations were applied to each crop based on N inputs and calculated for each diet and site scenario. These results were then compared with the results of N loss from DNDC. In all site and diet scenarios, DNDC predicted higher N losses compared with the assumed 4 % N loss from Beusen *et al* (2008). The N losses in the synthetic fertilizer scenario ranged from 0.007 kg PO_4 to 0.016 kg PO_4 equivalent per kg pig, with the highest N loss calculated from the home grown legume based diets. The estimated losses using the assumption from Beusen *et al* (2008) predicts losses ranging from 0.0004 to 0.0005 kg PO_4 equivalent in the synthetic fertilizer scenario. Estimates from DNDC are between 5 and 20 times higher than those using the assumptions of Beusen *et al* (2008).

Energy use for farm operations did not have a major impact on the total eutrophication potential. The NO_x emissions from fuel combustion has a relatively low impact, ranging from 0.01 % to 0.2 % of the total eutrophication potential in the synthetic fertilizer scenario and 0.2 % to 0.3 % of the total eutrophication potential in the slurry fertilizer scenario for all site and diet scenarios. The associated transport contributed 0.2 % to 0.9 % of the total eutrophication potential in the synthetic fertilizer scenario and 0.4 % to 2.0 % in the slurry fertilizer scenario. The contribution of stored slurry to the total eutrophication potential per kg pig in the slurry fertilizer scenarios is low, ranging from 0.8 % to 1.3 % for each site and diet scenario.

This LCA quantified the eutrophication potentials as PO_4 equivalents, however not all of the existing LCAs use the same equivalency value. The French LCA also used PO_4 equivalent and kg O_2 equivalent was used in the Sweden and kg NO_3 equivalent in the Danish studies. Again this highlights the difficulties in making comparisons between LCA results.

4.5.3 Acidification Associated with Pig Production

In the slurry fertilizer scenarios the acidification potential per kg pig is consistently higher when compared with the results in the synthetic fertilizer scenario. The major contributor to acidification potential is from stored slurry which is not included in the synthetic fertilizer scenario, followed by the production of fertilizers and pesticides. The associated acidification impacts associated with the transport of home grown feed ingredients to the processing plant and the long distance transport of soya.

In the synthetic fertilizer scenario the acidification potential ranges from 0.004 to 0.009 kg SO_4 equivalent. In the slurry fertilizer scenario, the acidification potential ranges from 0.0069 to 0.012 kg SO_4 equivalent. In each fertilizer scenario the lupin based diets have the lowest acidification potential and the soya based diets the highest acidification potential.

The lupin based diets have the lowest total acidification potential in both fertilizer scenarios. This is primarily due to the proportions of feed ingredients included in the diets, this includes the lowest inclusion of soybean meal (approximately half the amount compared with the pea and bean based diets). The acidification potential of soybean meal is the highest when compared with all feed ingredients, due to the higher fossil fuel requirements for the long distance transport. Consequently, soya based diets have the highest acidification potential in both fertilizer scenarios. This ranges from 0.01 kg SO_4 to 0.012 kg SO_4 equivalent in the slurry fertilizer scenario and 0.09 kg SO_4 equivalent in the synthetic fertilizer scenario for all sites.

For the combined production of fertilizers and pesticides in the synthetic fertilizer scenario, the contribution to total acidification potential ranges between 0.00061 and 0.00071 kg SO_4 (8.1 to 17.4 %), compared with the slurry fertilizer scenario which ranges between 0.00028 to 0.00033 kg SO_4 (2.8 % to 4.7 %). The energy required for farm operations had a slightly higher impact on the total acidification potential compared with the effects of fertilizer and pesticide production. This ranged from 0.00075 kg SO_4 to 0.00101 kg SO_4 (6.9 to 24.7 %) in the synthetic fertilizer scenario and 0.00081 kg SO_4 to 0.00107 kg SO_4 (6.2 % to 15.2 %) in the slurry fertilizer scenario.

SAA are used in all diets and are included to meet the amino acid requirements of the pig that cannot be met by the feed ingredients alone. The production of these SAA requires fossil energy and therefore contributes to the acidification potential. In both fertilizer scenarios the production of SAA contribute between 0.00046 kg SO_4 to 0.00080 kg SO_4, however when considering the percentage contribution to the total acidification potential per kg pig, this ranged between 8.1 and 20.1 % in the synthetic fertilizer scenario and 15.9 % and 47.4 % in the slurry fertilizer scenario. The requirements of SAA are higher in the lupin based diet due to the undesirable amino acid profile of lupins. In contrast, the soya based diets required the lower inclusions of SAA, due to the favourable amino acid profile of soya.

4.5.4 Advantages and Drawbacks of the LCA model

The described LCA is a useful tool to assess the environmental impacts of pig production systems. The LCA allows comparisons to be made between different diet scenarios within the same management system and therefore identifies specific diet scenarios which may lower environmental impact of pig production systems.

The LCA is constructed using several sub-models to assess individual components of the system. The outputs from each sub-model are linked into an integration model to finally calculate the total environmental impacts for the functional unit, 1 kg pig. The benefit of using sub-models to assess each component within the life cycle allows for a detailed assessment of individual components of the system and to determine where 'hot spots' or major contributors occur to the environmental impacts within a system.

This LCA uses a novel approach to calculate GHGs from crop growth by using DNDC. This allows; (1) quantification of GHGs on a daily basis, (2) the use of parameters adjusted to specific site conditions (3) the use of actual daily weather data for each site and (4) the impact of soil, climate and farm management factors to be explored. The outputs from DNDC therefore reflect the actual site and include the variations in natural conditions namely soil and climate. The GHGs associated with home grown legume production in some scenarios are higher than results which have previously been published for LCAs of pig production systems (Eriksson 2004; Basset-mens & Van der Werf 2005). Identified in this LCA, this is a result of including N_2O emissions from N fixation in legume crops. The LCAs which have used IPCC methodology do not account for N_2O losses from N fixed by legume crops.

The Animal Growth Model used in the LCA is also a useful approach with advantages such as determining accurate daily feed requirements of the pig allowing accurate calculations of feed consumed during the growth period. The model predicts daily N excretion and calculations of slurry production can be made.

The calculation of additional processes incorporated into the integration model are calculated from literature data and existing LCAs. The most appropriate data was implemented into the model to determine environmental impacts. However improvements could be made if actual data was available to quantify environmental impacts.

The LCA is particularly useful to calculate the environmental effects of different diet scenarios. To maximize the models ability, several more diets scenarios could be simulated to determine diets which have the lowest environmental impacts. As crop growth is the biggest contributor to total GWP per kg pig, a possible diet scenario could be modelled where the CP level is reduced and levels of SAA are increased to meet amino acid requirements, rather than CP *per se*. Additionally, the diets were formulated primarily to meet CP requirement and not on a least cost basis which is more commonly used in the commercial pig production. Therefore for future developments of the LCA, diets formulated on a least cost basis could also be included.

Difficulties however were encountered whilst sourcing site specific data, for example, soils. For UK soil types it was possible to determine reliable and accurate data from a reliable source for each site, however this was not the case for Brazilian soils. Consequently for Brazil, data was gathered from alternative sources and with less detail. As a result, when comparisons are made between DNDC results of UK crops and Brazilian soya, consideration must be taken of different accuracy of DNDC data input. The assumption was made that prior to the corn-soya rotation, land was previously used for arable crop production. However, if it had been assumed that the land was previously forest land, the results could potentially be very different. C pools occur in forest soils and during deforestation when soils are disturbed, C is released, which is then allocated to the newly cultivated crop, potentially increasing the GWP (Carey *et al*, 2001). This is a requirement of PAS 2050 and therefore if this LCA was to comply with PAS 2050 methodologies it would be a necessary requirement for the soya crop.

The GHGs calculated in this LCA for crop growth were generally higher compared to GHGs produced in the previously described LCAs for pig production discussed in Chapter 1. However, due to the use of the complex crop model, this indicates the potential benefits of modelling crop growth in more detail by taking into account the variations of site conditions.

4.6 Conclusion

The results were weighted from the lowest to highest results for each environmental impact category for each diet scenario at each site. These weighted values were then summed for each site to determine which diet scenario results with the highest and lowest environmental impacts. The conclusion is that the bean based diets have the lowest environmental impacts and the soya based diets have the highest environmental impact per kg pig. Both the pea and lupin based diets were concluded to have equal environmental impacts per kg pig.

Consideration was also made for each environmental impact category separately for both fertilizer scenarios. The bean based diets conclude to have the lowest GWP per kg pig and the soya based diets the highest GWP per kg. This differs for the acidification potential, as the pea based diets conclude with the lowest and the lupin based diets the highest acidification potential per kg pig. Finally with regards to eutrophication potential, the lupin based diets conclude the lowest and the soya based diets conclude the highest acidification potential per kg pig.

This work has highlighted, in environmental terms of pig production systems, that crop production is the main contributor. It has also been identified that the fertilizer scenario is important to consider when efforts are being made to reduce the environmental impacts within the management system. This includes (i) the importance for pig production systems to utilize slurry efficiently by considering crop nutrient requirements and (ii) the relevance of minimizing the amount of applied synthetic fertilizers.

This LCA has used a novel approach to model the environmental impacts of UK pig production systems. To allow detailed calculation of the environmental impacts associated with 1 kg pig by modelling specific sites within the UK. There may also be scope to lower the environmental impacts per kg pig by considering the environmental impacts of the feed ingredients used in diet formulations, whilst still meeting the nutrient requirements of the pig.

The overall conclusion is that UK legume based diets are associated with the lowest environmental impacts per kg pig when compared with a conventional soya based diet.

References

Abbasi, M.K., and Adams, W.A., 2000. Gaseous N emission during simultaneous nitrification-denitrification associated with mineral N fertilization to a grassland soil under field conditions. *Soil Biology and Biochemistry*. 32**,** 1251-1259.

Assimakopoulos, J.H., Kalivas, D.P., and Kollias, V.J., 2003. A GIS-based fuzzy classification for mapping the agricultural soils for N-fertilizers use. *Science of the Total Environment*. 309**,** 19-33.

British Atmospheric Data Centre. 2008. http://badc.nerc.ac.uk.

Baggs, E.M., Watson, C.A., and Rees, R.M., 2000. The fate of nitrogen from incorporated cover crop and green manure residues. *Nutrient Cycling in Agroecosystems*. 56, 153-163.

Basset-Mens, C., and van der Werf, H.M.G., 2005. Scenario-based environmental assessment of farming systems: the case of pig production in France. *Agriculture Ecosystems & Environment*. 105**,** 127-144.

Basset-Mens, C., Ledgard, S., and Boyes, M., 2009. Eco-efficiency of intensification scenarios for milk production in New Zealand. *Ecological Economics*. 68**,** 1615-1625.

Beusen, A.H.W., Bouwman, A.F., Heuberger, P.S.C., Van Drecht, G., and Van Der Hoek,K.W., 2008. Bottom-up uncertainty estimates of global ammonia emissions from global agricultural production systems. *Atmospheric Environment*. 42**,** 6067-6077.

Bouman, B.A.M., Jansen, H.G.P., Schipper, R.A., Nieuwenhuyse, A., Hengsdijk, H., and Bouma,J., 1999. A framework for integrated biophysical and economic land use analysis at different scales. *Agriculture Ecosystems & Environment*. 75**,** 55-73.

Brentrup, F., Küsters, J., Kuhlmann, H., and Lammel**,** J., 2004. Environmental impact assessment of agriculture production systems using the life cycle assessment methodology 1. Theorectical concept of a LCA method tailored to crop production. *European Journal of Agronomy*. 20, 247-264

Cabaraban, M.T.I., Khire, M.V., and Alocilja, E.C., 2008. Aerobic in-vessel composting versus bioreactor landfilling using life cycle inventory models. *Clean Technologies and Environmental Policy*. 10**,** 39-52.

Carey, E.V., Sala, A., Keane, R., and Callaway, R.M., 2001. Are old forests underestimated as global carbon sinks?. *Global Change Biology*. 7, 339-344

Chen, G.C., He, Z.L., Stoffella, P.J., Yang, X.E., Yu, S., and Calvert, D., 2006. Use of dolomite phosphate rock (DPR) fertilizers to reduce phosphorus leaching from sandy soil. *Environmental Pollution*. 139**,** 176-182.

Compassion in World Farming . www.ciwf.org.uk. 2009.

Courdier, R., Guerrin, F., Andriamasinoro, F., and Paillat, J.M., 2002. Agent-based simulation of complex systems: application to collective management of animal wastes. *Artificial Societies and Social Simulation*. 5, 30–56.

Da Silva, A.P. University of Sao Paulo, Department of soil science., 2009. Personal Communication

Dalgaard, R., Halberg, N., and Hermansen, J.E. Danish pork production: An environmental assessment. *DJF Animal science No*. 82. 2007. University of Aarhus, Faculty of agricultural sciences, Department of Agroecology and Environment, PO Box 50,DK-8830,Tjele.

Darnmer, K.H., Wollny, J., and Giebel, A., 2008. Estimation of the Leaf Area Index in cereal crops for variable rate fungicide spraying. *European Journal of Agronomy*. 28**,** 351-360.

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

Defra . Agricultural Survey. http://www.defra.gov.uk/esg/work_htm/publications/cs/farmstats_web/2_SURVEY_DATA_SEARCH/COUNTY_SIZE_GROUP_DATA/fd_tables_and_ah_tables/fd%20tables/2006fd/pdf/breeding_pigs_2006.pdf . 2006.

Defra. Defra RB209 fertiliser recommendations. 2008.

Dyer, J.A., and Desjardins, R.L., 2003. Simulated farm fieldwork, energy consumption and related greenhouse gas emissions in Canada. *Biosystems Engineering*. 85**,** 503-513.

Emmans, G.C., 1997. A method to predict the food intake of domestic animals from birth to maturity as a function of time. J*ournal of Theoretical Biology*. 186**,** 189-199.

Eriksson, I.S., Stern, S., and Nybrant, T., 2004. Environmental systems analysis of pig production: the impact of feed choice. *The International Journal of Life Cycle Assessment*. 10 (2), 143-154.

The Farm Management Handbook 2009. 30th edn. Edinburgh: Scottish Agriculture College.

Flechard, C.R., Ambus, P., Skiba, U., Rees, R.M., Hensen, A., van Amstel, A., Dasselaar, A.v.d.P., Soussana, J.F., Jones, M., Clifton-Brown, J., Raschi, A., Horvath, L., Neftel, A., Jocher, M., Ammann, C., Leifeld, J., Fuhrer, J., Calanca, P., Thalman, E., Pilegaard, K., Di Marco, C., Campbell, C., Nemitz, E., Hargreaves, K.J., Levy, P.E., Ball, B.C., Jones, S.K., van de Bulk, W.C.M., Groot, T., Blom, M., Domingues, R., Kasper, G., Allard, V., Ceschia, E., Cellier, P., Laville, P., Henault, C., Bizouard, F., Abdalla, M., Williams, M., Baronti, S., Berretti, F., and Grosz, B., 2007. Effects of climate and management intensity on nitrous oxide emissions in grassland systems across Europe. *Agriculture, Ecosystems & Environment*. 121**,** 135-152.

Freight Transportation Services., 2008. http://www.fta.co.uk/

Fumoto, T., Kobayashi, K., Li, C., Yagi, K., and Hasegawa, T., 2008. Revising a process-based biogeochemistry model (DNDC) to simulate methane emission from rice paddy fields under various residue management and fertilizer regimes. *Global Change Biology*. 14**,** 382-402.

Guafa, W., Peoples, M.B., Herridge, D.F., and Rerkasem, B., 1993. Nitrogen fixation, growth and yield of soybean grown under saturated soil culture and conventional irrigation. *Field Crops Research.* 32**,** 257-268.

Hansson, P.E., and Mattsson, B., 1999. Influence of derived operation-specific tractor emission data on results from an LCI on wheat production. *The International Journal of Life Cycle Assessment*. 4, 202-206.

Haynes, R.J., Martin, R.J., and Goh, K.M., 1993. Nitrogen fixation, accumulation of soil nitrogen and nitrogen balance for some field-grown legume crops. *Field Crops Research*. 35**,** 85-92.

Heinemann, A.B., Hoogenboom, G., and Chojnicki, B., 2002. The impact of potential errors in rainfall observation on the simulation of crop growth, development and yield. *Ecological Modelling*. 157**,** 1-21.

Herva, M., Franco, A., Ferreiro, S., Alvarez, A., and Roca, E., 2008. An approach for the application of the Ecological Footprint as environmental indicator in the textile sector. *Journal of Hazardous Materials*. 156**,** 478-487.

Huijbregts, M.A.J., and Seppala, J., 2001. Life cycle impact assessment of pollutants causing aquatic eutrophication. *International Journal of Life Cycle Assessment*. 6**,** 339-343.

International Organization of Standardization (ISO). www.iso.org

IPCC., 2001. Climate Change 2001: The Scientific Basis. Technical Summary of the Working Group I Report, Cambridge Univ. Press, New York.

IPCC., 2006. IPCC Guidelines for National Greenhouse Gas Inventories. Prepared by the National Greenhouse Gas Inventories Programme, IGES, Japan

IPCC., 2007. IPCC WG1 AR4 Report, Technical Summary, Climate Change 2007 – The Physical Science Basis Contribution of Working Group I to the 4th Assessment Report of the IPCC. Cambridge University Press, 33.

Janzen, H.H., Beauchemin, K.A., Bruinsma, Y., Campbell, C.A., Desjardins, R.L., Ellert, B.H., and Smith, E.G., 2003. The fate of nitrogen in agroecosystems: An illustration using Canadian estimates. *Nutrient Cycling in Agroecosystems*. 67**,** 85-102.

Johnson, A., 2009. British Quality Pigs. Personal Communication.

Kemanian, A.R., Stockle, C.O., Huggins, D.R., and Viega, L.M., 2007. A simple method to estimate harvest index in grain crops. *Field Crops Research*. 103**,** 208-216.

Kyriazakis, I., and Whittemore, C.T. Whittemore's science and practice of pig production. [Third edition]., 2006. Blackwell Publishing.

Lecoeur, J., and Sinclair, T.R., 2001. Harvest index increase during seed growth of field pea. *European Journal of Agronomy*. 14**,** 173-180.

Li, C., Frolking, S., and Frolking, T.A., 1992. A model of nitrous oxide evolution from soil driven by rainfall events: 1. Model structure and sensitivity. *Journal of Geophysical Research*. 97**,** 9759-9776.

Li, C.S., 2000. Modeling trace gas emissions from agricultural ecosystems. *Nutrient Cycling in Agroecosystems*. 58**,** 259-276.

Li, C.S. 2007a. Quantifying greenhouse gas emissions from soils: Scientific basis and modeling approach. *Soil Science and Plant Nutrition*. 53**,** 344-352.

Li, C.S. 2007b. User's Guide for the DNDC Model Institute for the Study of Earth, Oceans and Space. University of New Hampshire.

Lopez-Ridaura, S., van der Werf, H., Paillat, J.M., and Le Bris, B., 2009. Environmental evaluation of transfer and treatment of excess pig slurry by life cycle assessment. *Journal of Environmental Management*. 90**,** 1296-1304.

Liu, Y., Yu, Z., Chen, J., Zhang, F., Doluschitz, R., and Axmacher, J.C., 2006. Changes of organic carbon in an intensively cultivated agricultural region: A denitrification-decomposition (DNDC) modelling approach. *Science of the Total Environment*. 372**,** 203-214.

Mahmood, T., Ali, R., Malik, K.A., and Shamsi, S.R.A., 1998. Nitrous oxide emissions from an irrigated sandy-clay loam cropped to maize and wheat. *Biology and Fertility of Soils*. 27**,** 189-196.

Misselbrook, T.H., Van der Weerden, T.J., Pain, B.F., Jarvis, S.C., Chambers, B.J., Smith, K.A., Phillips, V.R., and Demmers, T.G.M., 2000. Ammonia emission factors for UK agriculture. *Atmospheric Environment*. 34**,** 871-880.

Niccolucci, V., Galli, A., Kitzes, J., Pulselli, R.M., Borsaa, S., and Marchettini, N., 2008. Ecological Footprint analysis applied to the production of two Italian wines. *Agriculture Ecosystems & Environment*. 128**,** 162-166.

NSRI . National Soil Research Institute. 2007.

PAS 2050., 2008. Specification for the assessment of the life cycle greenhouse gas emissions of goods and services. British Standards Institution, 389 Chiswick High Road, London W4 4AL.

Payraudeau, S., and van der Werf, H.M.G., 2005. Environmental impact assessment for a farming region: a review of methods. *Agriculture Ecosystems & Environment*. 107**,** 1-19.

Premier Nutrition Products. 2005. Ingredients Matrix, Premier Atlas 2005.

Rebitzer, G., Ekvall, T., Frischknecht, R., Hunkeler, D., Norris, G., Rydberg, T., Schmidt, W.-P., Suh, S., Weidema, B.P., and Pennington, D.W., 2004. Life cycle assessment: Part 1: Framework, goal and scope definition, inventory analysis, and applications. *Environment International*. 30**,** 701-720.

Rees, W.E., and Wackernagel, R.E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. *Ecological Economics.* 20, 3-24.

Reijs, J.W., Sonneveld, M.P.W., Srensen, P., Schils, R.L.M., Groot, J.C.J., and Lantinga, E.A., 2007. Effects of different diets on utilization of nitrogen from cattle slurry applied to grassland on a sandy soil in The Netherlands. *Agriculture, Ecosystems & Environment*. 118**,** 65-79.

Ringel, J., and Susenbeth, A., 2009. Lysine requirement for maintenance in growing pigs. *Livestock Science*. 120**,** 144-150.

Salo-vaananen, P.P., and Koivistoinen, P.E., 1996. Determination of protein in foods: comparison of net protein and crude protein values. *Food Chemistry*. 57**,** 27-31.

Sheldrick, W.F., and Lingard, J., 2004. The use of nutrient audits to determine nutrient balances in Africa. *Food Policy*. 29**,** 61-98.

Smith, P., Powlson, D.S., Glendining, M.J., and Smith, J.U., 1997. Potential for carbon sequestration in European soils: Preliminary estimates for five scenarios using results from long-term experiments. *Global Change Biology*. 3**,** 67-79.

Thomassen, M.A., 2008. Environmental Impact of Dairy Cattle Production Systems. PhD Thesis. Wageningen University. ISBN: 978-90-8504-891-6

Thornley, J. H. M. Grassland dynamics: an ecosystem simulation model. 1998. CAB International

Unkovich, M.J., and Pate, J.S., 2000. An appraisal of recent field measurements of symbiotic N2 fixation by annual legumes. *Field Crops Research*. 65, 211-228.

Vallejo, A., Skiba, U.M., Garcia-Torres, L., Arce, A., Lοπez-Fernανdez, S., and Sanchez-Martin, L., 2006. Nitrogen oxides emission from soils bearing a potato crop as influenced by fertilization with treated pig slurries and composts. *Soil Biology and Biochemistry*. 38, 2782-2793.

Van Belle, J.F., 2006. A model to estimate fossil CO2 emissions during the harvesting of forest residues for energy--with an application on the case of chipping. *Biomass and Bioenergy*. 30, 1067-1075.

Van den Bergh, J.C., and Verbruggen, H., 1999. Spatial sustainability, trade and indicators: an evaluation of the 'ecological footprint'. *Ecological Economics.* 29, 61-72.

Verge, X.P.C., Dyer, J.A., Desjardins, R.L., and Worth, D., 2007. Greenhouse gas emissions from the Canadian dairy industry in 2001. *Agricultural Systems*. 94, 683-693.

Verge, X.P.C., Dyer, J.A., Desjardins, R.L., and Worth, D., 2008. Greenhouse gas emissions from the Canadian beef industry. *Agricultural Systems*. 98, 126-134.

Verge, X.P.C., Dyer, J.A., Desjardins, R.L., and Worth, D., 2009. Greenhouse gas emissions from the Canadian pork industry. *Livestock Science.* 121, 92-101.

Vink, E.T.H., Rabago, K.R., Glassner, D.A., and Gruber, P.R., 2003. Applications of life cycle assessment to NatureWorks (TM) polylactide (PLA) production. *Polymer Degradation and Stability*. 80, 403-419.

Wackernagel, M., and Rees, W.E., 1997. Perceptual and structural barriers to investing in natural capital: Economics from an ecological footprint perspective. *Ecological Economics.* 20**,** 3-24.

Wellock, I.J., Emmans, G.C., and Kyriazakis, I., 2003. Modelling the effects of thermal environment and dietary composition on pig performance: model testing and evaluation. *Animal Science*. 77**,** 267-276.

Wellock, I.J., Emmans, G.C., and Kyriazakis, I., 2004. Describing and predicting potential growth in the pig. *Animal Science.* 78**,** 379-388.

Williams, A.G., Audsley, E., and Sandars, D.L., 2006. Determining the environmental burdens and resource use in the production of agricultural and horticultural commodities. Main report. Defra Research Project IS0205. Bedford: Cranfield University and Defra. Available on www.silsoe.cranfield.ac.uk, and www.defra.gov.uk.

Yan, X., 2009. Energy demand and greenhouse gas emissions during the production of a passenger car in China. *Energy Conversion and Management*. 50**,** 2964-2966.

Zhong, Z., Lemke, R.L., and Nelson, L.M., 2009. Nitrous oxide emissions associated with nitrogen fixation by grain legumes. *Soil Biology and Biochemistry*. 41**,** 2283-2291.

Appendices

Appendix A

York2007.txt - Notepad

File Edit Format View Help

York2007

1	9.9	3.2	0.61
2	6	2.7	0.61
3	9.8	1.9	1
4	10.5	6.8	0.15
5	9	6.1	1.2
6	9.4	4.5	0.06
7	8.8	5.5	0.21
8	9.5	4.5	1.53
9	11.2	5.2	1.63
10	10.3	2.1	5.57
11	9.9	3	3.65
12	10	4.6	4.74
13	11	6	0.67
14	9.6	4.7	0.42
15	8	5	1.62
16	9.1	0	0.01
17	7.3	-0.2	0.42
18	9	-0.2	1.4
19	9	3	2.72
20	11.1	2.5	0.5
21	5.6	2	3.18
22	4.6	-0.4	1.8
23	4.5	-2	0.27
24	2.6	-2.1	0
25	3.7	-4.4	0
26	5	-2.8	0.16
27	8.4	1.8	0.08
28	7.5	4.1	0
29	9.1	6.4	0
30	9.7	2.4	0
31	7.8	1.2	0
32	10	5.6	0

Figure A1 Example of a weather file used as an input into DNDC.

Appendix B

Table B1 The percentage contribution of total GHG per kg feed ingredient in East Anglia in the synthetic fertilizer scenario.

	CO_2 (%)	**N_2O (%)**	**GWP (kg CO_2 equivalent^{100})**
Beans	51.4	48.6	0.45
Peas	66.7	33.3	0.63
Lupins	65.7	34.3	0.65
Spring Barley	48.6	51.4	0.83
Winter Barley	60.7	39.3	1.08
Wheat	86.5	13.5	0.29
Wheatfeed	86.5	13.5	0.15
Rapeseed	23.3	76.7	1.06
Soya	0.0	100.0	1.59

Table B2 The percentage contribution of total GHG per kg feed ingredient in Yorkshire in the synthetic fertilizer scenario.

	CO_2 (%)	**N_2O (%)**	**GWP (kg CO_2 equivalent^{100})**
Beans	41.1	58.9	0.20
Peas	30.0	70.0	0.16
Lupins	43.0	57.0	0.22
Spring Barley	58.5	41.5	1.11
Winter Barley	68.0	32.0	1.35
Wheat	73.1	26.9	0.41
Wheatfeed	73.1	26.9	0.20
Rapeseed	0.0	100.0	0.64
Soya	0.0	100.0	1.59

Table B3 The percentage contribution of total GHG per kg feed ingredient for silty clay loam in East Anglia in the synthetic fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent^{100})
Beans	64.7	35.3	0.89
Peas	65.2	34.8	0.88
Lupins	64.5	35.5	0.90
Spring Barley	46.5	53.5	1.05
Winter Barley	68.1	31.9	1.24
Wheat	80.3	19.7	0.35
Wheatfeed	80.3	19.7	0.18
Rapeseed	34.6	65.4	1.21
Soya	0.0	100.0	1.59

Table B4 The percentage contribution of total GHG per kg feed ingredient for silty clay loam in Yorkshire in the synthetic fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent^{100})
Beans	10.7	89.3	0.14
Peas	0.0	100.0	0.10
Lupins	21.6	78.4	0.17
Spring Barley	66.6	33.4	1.04
Winter Barley	73.9	26.1	1.27
Wheat	71.5	28.5	0.41
Wheatfeed	71.5	28.5	0.21
Rapeseed	0.0	100.0	0.44
Soya	0.0	100.0	1.59

Table B5 The percentage contribution of total GHG per kg feed ingredient in East Anglia in the slurry fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent^{100})
Beans	49.9	50.1	0.55
Peas	50.7	49.3	0.54
Lupins	49.6	50.4	0.55
Spring Barley	39.1	60.9	0.89
Winter Barley	19.5	80.5	1.13
Wheat	85.8	14.2	0.37
Wheatfeed	85.8	14.2	0.19
Rapeseed	0.0	100.0	0.92
Soya	0.0	100.0	1.59

Table B6 The percentage contribution of total GHG per kg feed ingredient in Yorkshire in the slurry fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent^{100})
Beans	61.0	39.0	0.35
Peas	62.7	37.3	0.35
Lupins	60.2	39.8	0.36
Spring Barley	50.7	49.3	1.14
Winter Barley	42.7	57.3	1.30
Wheat	70.0	30.0	0.51
Wheatfeed	70.0	30.0	0.26
Rapeseed	0.0	100.0	0.92
Soya	0.0	100.0	1.59

Table B7 The percentage contribution of total GHG per kg feed ingredient for silty clay loam in East Anglia in the slurry fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent[100])
Beans	48.5	51.5	0.71
Peas	49.1	50.9	0.71
Lupins	47.3	52.7	0.73
Spring Barley	37.8	62.2	1.16
Winter Barley	36.7	63.3	1.23
Wheat	81.3	18.7	0.44
Wheatfeed	81.3	18.7	0.22
Rapeseed	16.4	83.6	1.17
Soya	0.0	100.0	1.59

Table B8 The percentage contribution of total GHG per kg feed ingredient for silty clay loam in Yorkshire in the slurry fertilizer scenario.

	CO_2 (%)	N_2O (%)	GWP (kg CO_2 equivalent[100])
Beans	60.3	39.7	0.34
Peas	61.6	38.4	0.34
Lupins	59.9	40.1	0.35
Spring Barley	56.4	43.6	1.06
Winter Barley	42.0	58.0	1.26
Wheat	69.3	30.7	0.51
Wheatfeed	69.3	30.7	0.25
Rapeseed	0.0	100.0	0.39
Soya	0.0	100.0	1.59

Appendix C

Table C1 The percentage contribution of each component of the pig production system to total GWP per kg pig in the synthetic fertilizer scenario.

	GWP (kg CO_2 equivalent^{100})	Crop Growth	Farm Operations	Fertilizer and Pesticide Production	Transport	Drying	Extraction	Energy for Buildings	Manure management	Enteric CH_4
EA Beans	2.03	70.3	5.8	10.9	1.6	1.7	2.2	7.3	0.0	0.3
EA Peas	2.47	74.8	5.2	8.9	1.5	1.5	1.9	6.0	0.0	0.2
EA Lupins	2.22	71.6	5.9	11.2	0.9	1.8	1.6	6.7	0.0	0.3
EA Soya	2.52	71.4	4.1	10.0	3.4	1.6	3.3	5.9	0.0	0.2
Y Beans	1.93	68.9	6.4	11.0	1.6	1.7	2.3	7.7	0.0	0.3
Y Peas	2.18	71.0	6.4	10.1	1.7	1.6	2.2	6.8	0.0	0.3
Y Lupins	1.99	68.5	6.8	12.2	1.0	1.9	1.8	7.5	0.0	0.3
Y Soya	2.67	72.9	4.3	9.4	3.2	1.2	3.2	5.6	0.0	0.2
SCL EA Beans	2.43	75.5	4.7	8.9	1.3	1.4	1.8	6.1	0.0	0.2
SCL EA Peas	2.86	78.5	4.4	7.6	1.3	1.2	1.6	5.2	0.0	0.2
SCL EA Lupins	2.54	75.6	5.0	9.6	0.8	1.5	1.4	5.8	0.0	0.2
SCL EA Soya	2.73	73.9	3.7	9.0	3.1	1.5	3.1	5.4	0.0	0.2
SCL Y Beans	1.85	67.4	6.6	11.7	1.7	1.8	2.4	8.0	0.0	0.3
SCL Y Peas	2.07	69.4	6.7	10.6	1.8	1.7	2.3	7.2	0.0	0.3
SCL Y Lupins	1.92	67.4	7.1	12.6	1.1	1.9	1.9	7.8	0.0	0.3
SCL Y Soya	2.61	72.0	4.4	9.6	3.3	1.5	3.3	5.7	0.0	0.2

Table C2 The percentage contribution of each component of the pig production system to total GWP per kg pig in the slurry fertilizer scenario.

	GWP (kg CO_2 equivalent^{100})	Crop Growth	Farm Operations	Fertilizer and Pesticide Production	Transport	Drying	Extraction	Energy for Buildings	Manure management	Enteric CH4
EA Beans	2.40	65.1	4.8	3.9	1.3	1.5	1.8	6.2	15.1	0.2
EA Peas	2.63	68.5	4.9	3.1	1.4	1.4	1.8	5.7	13.1	0.2
EA Lupins	2.45	65.3	5.3	4.3	0.8	1.6	1.4	6.1	14.9	0.2
EA Soya	2.80	66.8	3.7	3.7	3.1	1.4	3.0	5.3	12.8	0.2
Y Beans	2.45	64.0	7.3	3.8	1.3	1.4	1.9	6.1	14.0	0.2
Y Peas	2.66	67.8	5.2	4.1	1.4	1.3	1.8	5.6	12.6	0.2
Y Lupins	2.47	66.0	5.5	4.2	0.8	1.5	1.5	6.0	14.2	0.2
Y Soya	3.00	69.4	3.8	3.4	2.9	1.3	2.9	5.0	11.1	0.2
SCL EA Beans	2.67	68.7	4.3	3.5	1.2	1.3	1.6	5.6	13.6	0.2
SCL EA Peas	3.07	71.2	4.1	3.1	1.2	1.2	1.5	4.8	12.7	0.2
SCL EA Lupins	2.79	69.7	4.6	3.8	0.7	1.4	1.2	5.3	13.1	0.2
SCL EA Soya	3.08	70.0	3.3	3.3	2.8	1.3	2.7	4.8	11.6	0.2
SCL Y Beans	2.29	63.9	5.3	4.2	1.4	1.5	2.0	6.5	15.0	0.3
SCL Y Peas	2.47	65.7	5.6	4.0	1.5	1.4	1.9	6.0	13.6	0.2
SCL Y Lupins	2.41	65.4	5.7	4.3	0.8	1.5	1.5	6.2	14.3	0.2
SCL Y Soya	2.95	68.9	3.9	3.4	2.9	1.3	2.9	5.0	11.3	0.2

Appendix D

Table D1 The percentage contribution of each component of the pig production system to total eutrophication potential per kg pig in the synthetic fertilizer scenario.

	Eutrophication (kg PO_4 equivalent)	Stored Slurry	N leached	Farm operations	Transport	Pesticide and fertilser production	P Loss from slurry fertilizer
EA Beans	0.108	0.0	6.3	0.1	0.4	92.9	0.2
EA Peas	0.103	0.0	6.5	0.2	0.5	92.7	0.3
EA Lupins	0.122	0.0	6.4	0.1	0.2	93.0	0.2
EA Soya	0.122	0.0	5.5	0.1	0.9	93.3	0.2
Y Beans	0.116	0.0	13.5	0.1	0.4	85.8	0.2
Y Peas	0.111	0.0	13.1	0.2	0.4	86.1	0.2
Y Lupins	0.127	0.0	9.8	0.1	0.2	89.6	0.2
Y Soya	0.125	0.0	8.0	0.1	0.9	90.8	0.2
SCL EA Beans	0.109	0.0	8.0	0.1	0.4	91.3	0.2
SCL EA Peas	0.105	0.0	8.1	0.1	0.5	91.1	0.3
SCL EA Lupins	0.124	0.0	8.0	0.1	0.2	91.4	0.2
SCL EA Soya	0.123	0.0	6.4	0.1	0.9	92.4	0.2
SCL Y Beans	0.116	0.0	13.3	0.1	0.4	85.9	0.2
SCL Y Peas	0.111	0.0	13.0	0.2	0.4	86.2	0.2
SCL Y Lupins	0.133	0.0	13.8	0.1	0.2	85.7	0.2
SCL Y Soya	0.129	0.0	11.0	0.1	0.9	87.8	0.2

Table D2 The percentage contribution of each component of the pig production system to total eutrophication potential per kg pig in the slurry fertilizer scenario.

	Eutrophication (kg PO_4 equivalent)	Stored Slurry	N leached	Farm operations	Transport	Pesticide and fertilser production	P Loss from slurry fertilizer
EA Beans	0.050	1.2	14.6	0.3	0.9	82.5	0.5
EA Peas	0.049	1.2	14.6	0.3	1.0	82.4	0.6
EA Lupins	0.056	1.1	14.7	0.3	0.5	82.9	0.5
EA Soya	0.054	1.1	13.0	0.2	2.2	83.1	0.4
Y Beans	0.061	0.9	30.7	0.3	0.7	66.9	0.4
Y Peas	0.059	0.9	29.1	0.3	0.8	68.3	0.5
Y Lupins	0.070	0.8	31.6	0.3	0.4	66.5	0.4
Y Soya	0.064	0.9	27.1	0.2	1.8	69.6	0.4
SCL EA Beans	0.051	1.2	17.3	0.3	0.8	79.9	0.5
SCL EA Peas	0.051	1.3	17.1	0.3	1.0	79.8	0.5
SCL EA Lupins	0.059	1.0	18.0	0.3	0.5	79.7	0.5
SCL EA Soya	0.055	1.1	14.9	0.2	2.1	81.3	0.4
SCL Y Beans	0.060	0.4	29.6	0.3	0.7	68.6	0.4
SCL Y Peas	0.059	1.0	28.7	0.3	0.8	68.7	0.5
SCL Y Lupins	0.070	0.8	31.0	0.3	0.4	67.1	0.4
SCL Y Soya	0.064	0.9	26.6	0.2	1.8	69.9	0.6

Appendix E

Table E1 The percentage contribution of each component of the pig production system to total acidification potential per kg pig in the synthetic fertilizer scenario.

	Acidification (kg SO_2 equivalent)	Stored slurry	Farm operations	Transport	Pesticide and fertilizer production	SAA production
EA Beans	0.005	0.0	17.7	52.4	13.6	16.3
EA Peas	0.005	0.0	18.3	59.1	12.9	9.7
EA Lupins	0.004	0.0	22.8	39.8	17.4	19.9
EA Soya	0.009	0.0	8.3	75.5	8.1	8.1
Y Beans	0.005	0.0	18.9	51.6	13.4	16.0
Y Peas	0.005	0.0	20.5	57.3	12.8	9.4
Y Lupins	0.004	0.0	24.7	38.9	16.9	19.5
Y Soya	0.009	0.0	8.6	75.2	8.1	8.1
SCL EA Beans	0.005	0.0	16.7	53.2	13.6	16.5
SCL EA Peas	0.005	0.0	17.7	59.6	12.9	9.8
SCL EA Lupins	0.004	0.0	22.5	40.2	17.2	20.1
SCL EA Soya	0.009	0.0	6.9	76.7	8.1	8.2
SCL Y Beans	0.005	0.0	18.9	51.7	13.4	16.1
SCL Y Peas	0.005	0.0	20.5	57.3	12.8	9.4
SCL Y Lupins	0.004	0.0	24.6	39.0	16.8	19.5
SCL Y Soya	0.009	0.0	8.5	75.3	8.1	8.1

Table E2 The percentage contribution of each component of the pig production system to total acidification potential per kg pig in the slurry fertilizer scenario.

	Acidification (kg SO_2 equivalent)	Stored slurry	Farm operations	Transport	Pesticide and fertilizer production	SAA production
EA Beans	0.0076	42.9	10.8	32.3	3.9	10.1
EA Peas	0.0076	41.0	12.0	37.1	3.8	6.1
EA Lupins	0.0069	47.2	13.6	23.0	4.7	11.5
EA Soya	0.0115	27.8	6.2	57.0	2.8	6.1
Y Beans	0.0075	41.0	12.5	32.5	3.9	10.1
Y Peas	0.0076	39.4	13.8	36.9	3.8	6.1
Y Lupins	0.0069	45.6	15.1	23.1	4.7	11.6
Y Soya	0.0115	26.2	7.6	57.3	2.9	6.1
SCL EA Beans	0.0076	43.2	10.7	32.3	3.8	10.0
SCL EA Peas	0.0079	44.0	11.1	35.4	3.6	5.8
SCL EA Lupins	0.0069	47.4	13.3	23.1	4.6	11.6
SCL EA Soya	0.0115	27.9	6.2	57.1	2.8	6.1
SCL Y Beans	0.0075	41.0	12.4	32.5	3.9	10.1
SCL Y Peas	0.0076	39.4	14.0	36.8	3.8	6.0
SCL Y Lupins	0.0068	45.0	15.2	23.3	4.7	11.7
SCL Y Soya	0.0101	15.9	8.6	65.3	3.3	7.0

Appendix F

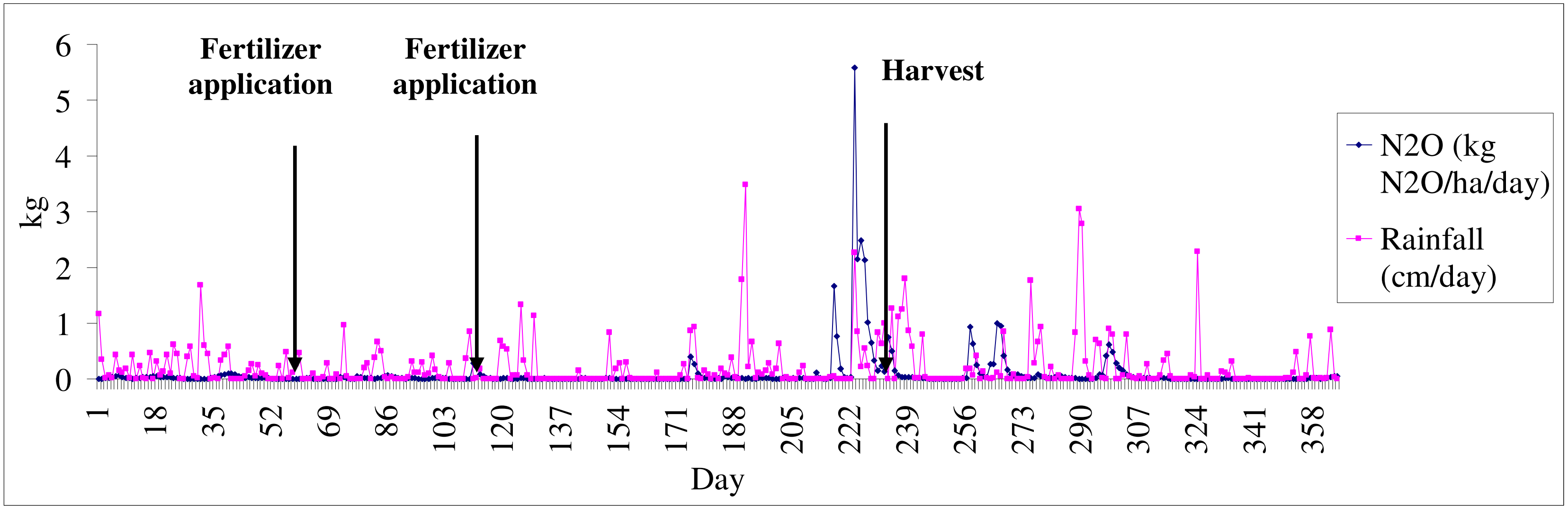

Figure F1 N_2O (kg/ha/day) released from soil and rainfall (cm) during the growth of winter barley in the Yorkshire bean rotation.

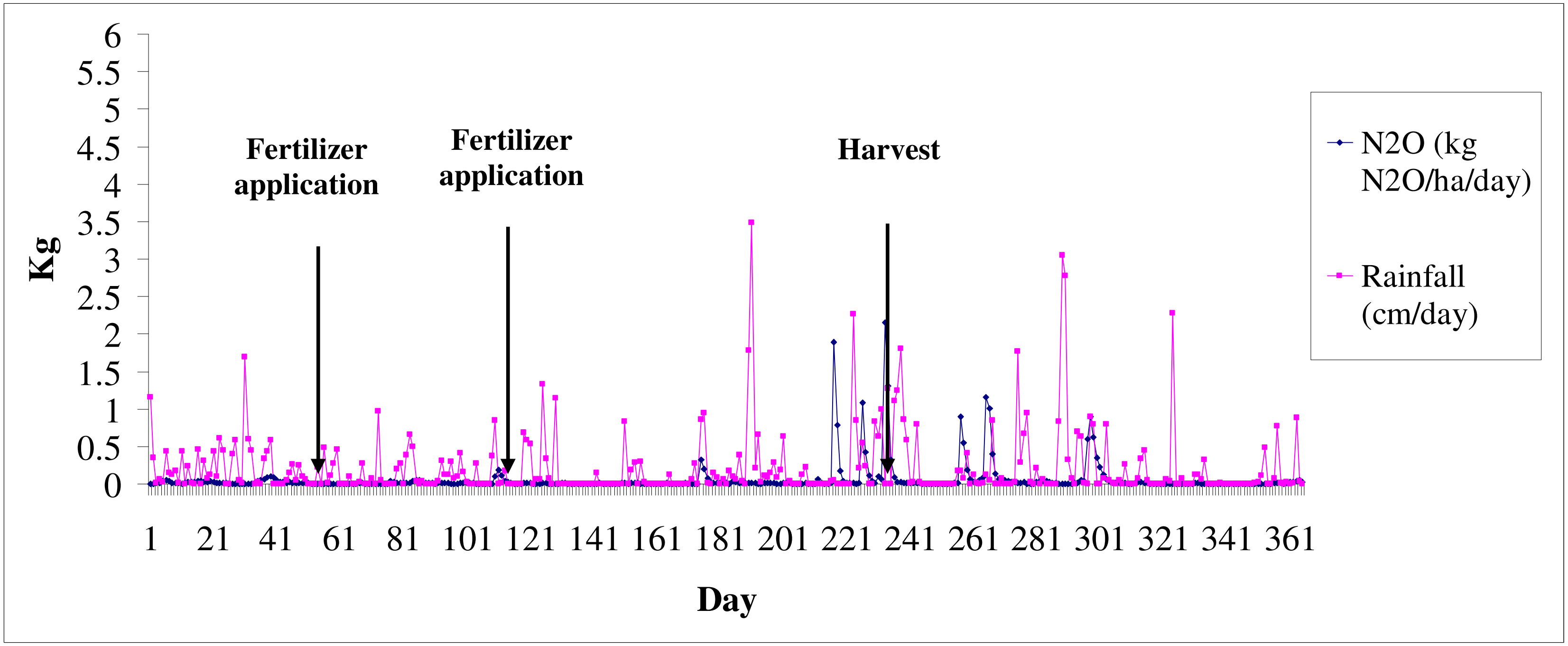

Figure F2 N_2O (kg/ha/day) released from soil and rainfall (cm) after the high rainfall events have been reduced to 0 at days 223 and 224 during the growth of winter barley in the Yorkshire bean rotation.

List of Publications

Stephen, K.L., Tolkamp, B.J., Topp, C.F.E., Houdijk, J.G.M. and Kyriazakis, I. (2009). Environmental impacts of UK pig production systems: Analysis using Life Cycle Assessment. *Aspects of Applied Biology, 93, Integrated Agricultural Systems: Methodologies, Modelling and Measuring, published by the Association of Applied Biologists, The Warwick Enterprise Park, Wellesbourne, Warwick, CV35 9EF, UK, pp33 – 39.*