A Thesis Submitted for the Degree of Doctor of Philosophy at

Harper Adams University

THE LIFE CYCLE GREENHOUSE GAS EMISSIONS OF RENDERED PRODUCTS

by

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Abstract

The main research objectives were: (i) to quantify the greenhouse gas (GHG) emissions of rendered products, and (ii) to evaluate the effect of the inclusion of ingredients derived from terrestrial animal by-products (ABP) on the GHG emissions of animal diets. Generic life cycle assessment methodology was used to study the main systems: category 1 and 3 mammalian rendering, on-farm broiler production, chicken meat processing, poultry rendering, and salmon feed production. UK industry data were collected to build the life cycle inventories. The effect of fuels used (natural gas (NG) and rendered fat (RF)) in the rendering industry and alternative co-product handling approaches were investigated.

GHG emissions calculated were −0.77 and 0.15 kg CO_2e/kg category 1 and 3 mammalian rendered fat respectively and 0.15 kg CO_2e/kg mammalian processed animal protein (PAP) for the mean proportion of NG and RF. GHG emissions were 1.798 and 1.901 kgCO_2e/kg live weight for 'Standard' and 'Heavy' broiler production systems respectively. GHG emissions were 3.415, 2.042, 3.495, and 3.257 kgCO_2e/kg chicken meat using economic allocation, mass allocation, main product, and system expansion respectively. GHG emissions of poultry PAP were 0.325 and 1.201, 7.555 and 8.423, -0.178 and 0.698 kg CO_2/kg for economic allocation, mass allocation, main product employed to partition between chicken meat and poultry ABP and for RF and NG respectively. The inclusion of poultry PAP instead of fish meal derived from reduction fisheries in salmon feed production resulted in higher and lower GHG emissions when employing mass and economic allocation respectively.

Economic allocation is an adequate co-product handling method for animal by-product systems because the driver for their production is the demand for the main commodity (edible meat products). The GHG emissions of rendered products were similar or low relative to marginal products such as palm oil and soya bean meal because (i) ABP have a low or null value and therefore incur low or zero emissions from their production, (ii) the rendering process produces biofuels that are used to offset the use of fossil fuels, and (iii) palm oil and soya bean incur emissions from agriculture and land transformation.

Declaration

I declare that the work in this thesis is my own and has not been submitted or accepted in any previous application for a degree or a diploma in any academic institution.

Angel D Ramirez

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Dedication

This work is dedicated to my parents and my aunt.

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List of abbreviations

ABP	Animal by-product
AD	Anaerobic digestion
BOD	Biological oxygen demand
BSE	Bovine spongiform encephalopathy
BSI	British Standards Institution
C	Carbon
CF	Carbon footprint
CFC	chlorofluorocarbon
CH_4	Methane
CHP	Cogeneration of heat and power
CO_2	Carbon dioxide
CO_2e	Carbon dioxide equivalent
COD	Chemical oxygen demand
CW	Carcass weight
DDGS	Dark distilled grains
DECC	Department of Energy and Climate Change
DEFRA	Department of Environment, Food and Rural Affairs
DM	dry matter
DRP	Dry rendered product
e.g.	For example
EA	Economic allocation
EBLEX	English Beef and Lamb Executive
EC	European Commission
EPD	Environmental product declarations
EPS	Environmental Priority Strategies (in Product Design)
et al.	And others
etc.	Etcetera
EU	European Union
FABRA	Food chain and Biomass Renewables Association Limited
FAO	Food and Agriculture Organization (of the United Nations)
FBC	Fluidised bed combustion
FCR	Feed conversion ratio
FU	Functional unit
GHG	Greenhouse gas
GHV	Gross Heating Value
GWP	Global Warming Potential
HCFC	hydroclorofluorocarbons
HCL	Hydrogen chloride

HFM	Hydrolysed feather meal
i.e.	that is
ICE	Internal combustion engine
IPCC	Intergovernmental Panel on Climate Change
ISO	International Organisation for Standardisation
KoP	Killing out percentage
LCA	Life cycle assessment
LPG	Liquefied petroleum gas
LULUCF	Land use, land use change and forestry
LW	Live weight
MA	Mass allocation
MBM	Meat and bone meal
MP	Main product
MRF	Mammalian rendered fat
N	Nitrogen
N_2O	Nitrous oxide
NH_3	Ammonia
NHV	Net Heating Value
NO_x	Nitrogen oxides
NREL	National Renewable Energy Laboratory (of the United States of America)
ODS	Ozone-depleting substances
OECD	Organisation for Economic Co-operation and Development
P	Phosphorus
PAP	Processed animal protein
PAS	Publicly available specification
PPAP	Poultry processed animal protein
PRF	Poultry rendered fat
RO	Renewable Obligation
RSPCA	The Royal Society for the Prevention of Cruelty to Animals
SE	System expansion
SETAC	Society of Environmental Toxicology and Chemistry
SO_2	Sulphur dioxide
SRM	Specified risk material
TSE	Transmissible Spongiform Encephalopathy
UK	United Kingdom
UKRA	United Kingdom Renderers Association
UN	United Nations
US	United States of America
vCJD	variant Creutzfeld-Jacob Disease

Chapter 1

Introduction

1 Introduction

Climate change represents a serious threat to human society and the natural environment. During the last century the Earth warmed approximately 0.7 °C largely due to increased greenhouse gas (GHG) emissions resulting from human activity. Without action to reduce GHG emissions global temperatures are predicted to rise 0.3 – 6.4 °C by 2100, with sea levels rising by 0.18 – 0.59 metres compared to 1990 levels (Solomon *et al.*, 2007a). This is likely to result in more extreme weather occurrences (floods, drought), with developing countries becoming more at risk from disease, hunger and famine. To combat climate change international and national policy has been developed to reduce GHG emissions. For example, the Kyoto protocol (UN, 1998) is an international agreement to reduce emissions of six GHGs, including carbon dioxide, nitrous oxide and methane. Under the agreement the European Union (EU) has agreed to collectively reduce emissions by 8.5% and the UK has specifically agreed to reduce emissions by 12.5% below 1990 levels by 2012 (UK, 2008a).

Life Cycle Assessment (LCA) is a tool that is used to investigate and quantify the environmental impact of a given product or service during its life cycle (ISO, 2006b; a) . Examples of environmental impacts that can be assessed include climate change, acidification, smog, ozone layer depletion, eutrophication, human toxicological pollutants, desertification and depletion of minerals and fossil fuels. The life cycle of a product or service can be considered as a series of consecutive or interlinked stages from extraction of the raw materials to disposal of the end products. The LCA methodology can also be used to analyse specific environmental impacts (e.g. Climate Change) or specific stages of a product or services life cycle (e.g. production).

Animal by-products (ABP) are unavoidable by-products of the livestock production system. In the EU, 17 million tonnes of slaughterhouse by-product are produced every year (Woodgate and van der Veen, 2004). The rendering industry is accepted to play an important role in the food chain by transforming ABP into processed proteins and rendered fats, which are used as inputs for other industries. Rendering involves physical transformation, with the processes involving particle size reduction, heat treatment (for dehydration and microbial sterilization), pressing, separation and milling. Feedstock for the rendering industry consists primarily of slaughterhouse by-products, but also on-farm fallen stock and butchery and supermarket wastes. Rendering plants deal with different categories of raw materials (EC, 2002; 2009) and the products of rendering, are either burnt as fuel (category 1), or used in the fertiliser, pet food and chemical industries (category 3). Alternative sources of fats and proteins are palm oil and soybean meal. To date there have been no LCA or "carbon footprint" studies conducted on the UK rendering industry. Consequently the GHG emissions of rendered products have not been clearly defined.

The main objective of this project was to determine the GHG emissions associated with the production of rendered products in the UK. Under current legislation category 3 proteinaceous material can be included as pet food ingredients. However, the European Commission is considering to allow the feeding of poultry derived processed animal protein to farmed pig and farmed fish and pig derived processed animal protein to farmed birds and farmed fish (fish derived material is already allowed to be used in the feeding of non-ruminants) (Gleadle, 2011). Therefore it is expected that poultry derived processed animal proteins will be allowed to be included in farmed fish feeds in the short to midterm. In the UK, farmed fish feeds are normally formulated with fishmeal as the predominant protein source. A second objective of the project was to undertake comparative LCA of GHG emissions of salmonid feeds production based on either vegetable, fishmeal or terrestrial ABP derived ingredients as the predominant protein source.

Chapter 2

Literature review: Introduction to animal by-products and life cycle assessment

2 Literature review: Introduction to animal by-products and life cycle assessment

2.1 The animal by-product system

The animal by-product (ABP) system includes animal production, slaughtering, meat processing, meat consumption, ABP processing, and ABP use or disposal (Figure 1). Each of these elements of the ABP system is described in the following sections, with a focus on ABP processing and rendering (the focus of the research presented).

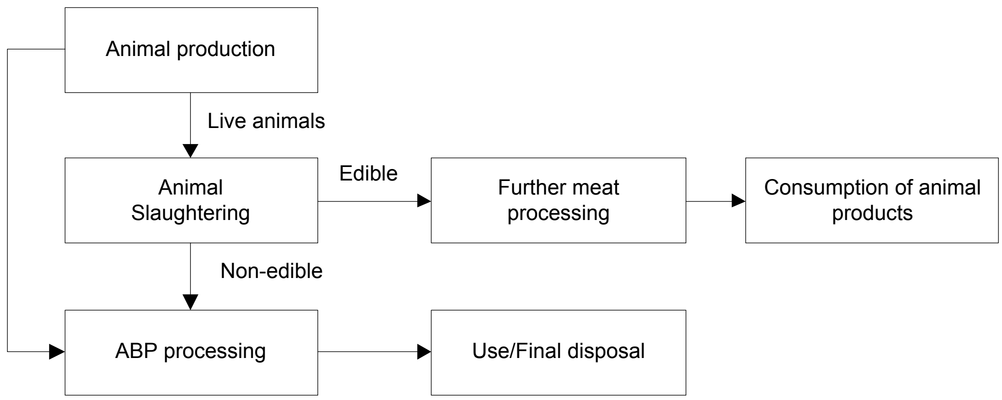

Figure 1 The animal by-product system

2.1.1 Meat production trend

The principal purpose of animal agriculture is to produce animal derived foods such as meat products, milk and eggs. Meat products are mainly produced from beef cattle, lamb, pigs and poultry. Globally meat production (including all different animals) increased steadily between 2000 and 2009 from approximately 233.5 to 283.9 million tonnes per annum (p.a.), corresponding to a 21.6 % increase (Figure 2). During the same period total European meat production increased from approximately 51.7 to 54.8 million tonnes p.a. (representing a 6% increase), whilst UK production remained constant at 3.4 million tonnes per annum. Globally, meat production is projected to more than double from 1999-2001 to 2050 (229 to 465 million tonnes), with poultry being the commodity of choice (Steinfeld *et al.*, 2006).

Poultry represents the largest UK meat production sector. In 2009, 1,652 thousand tonnes of poultry meat was produced (Figure 3), of which 88.6 % was chicken meat. Chicken meat production increased from 1,214 to 1,463 thousand tonnes between 2000 and 2009 (a 20% increase). Chicken meat increased its contribution to meat production from 34.6% to 41.4% between 2000 and 2009. In 2000, the second and third largest amount of meat produced in the UK were pork and beef, with 899 and 705 thousand tonnes produced, respectively. However, in 2004, beef production surpassed pig meat production. In 2009, 849.9 and 720.3 thousand tonnes of beef and pork were produced, respectively. Lamb production decreased from 383.0 thousand tonnes in 2000 to 302.6 in 2009 (Figure 3). Between 2000 and 2009, UK meat production was steady (Figure 2) however a shift in the contribution is observed. In concordance with global trends, meat produced from mammalian animals has decreased, whilst meat production from birds (mostly broiler chicken) has increased.

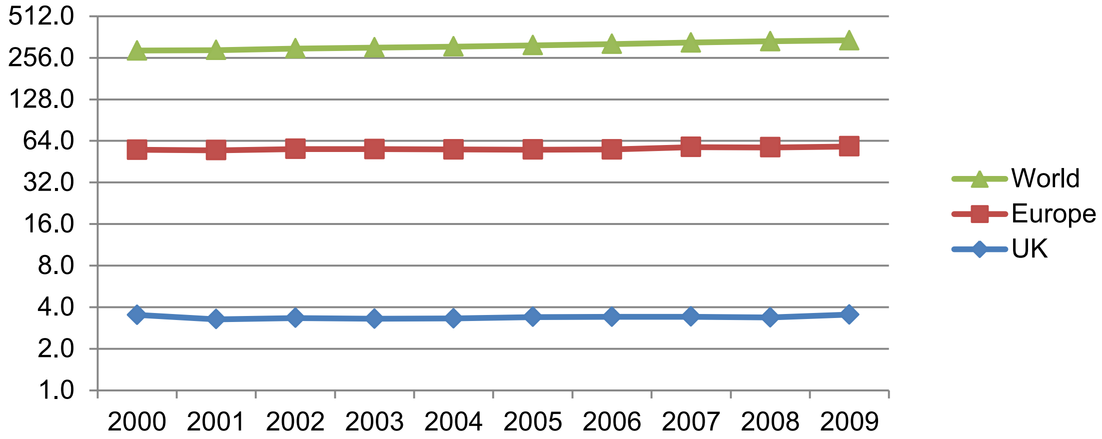

Figure 2 World, Europe and United Kingdom total meat production in million tonnes between 2000 and 2009 (FAO, 2011). Quantity axis is in logarithmic base 2.

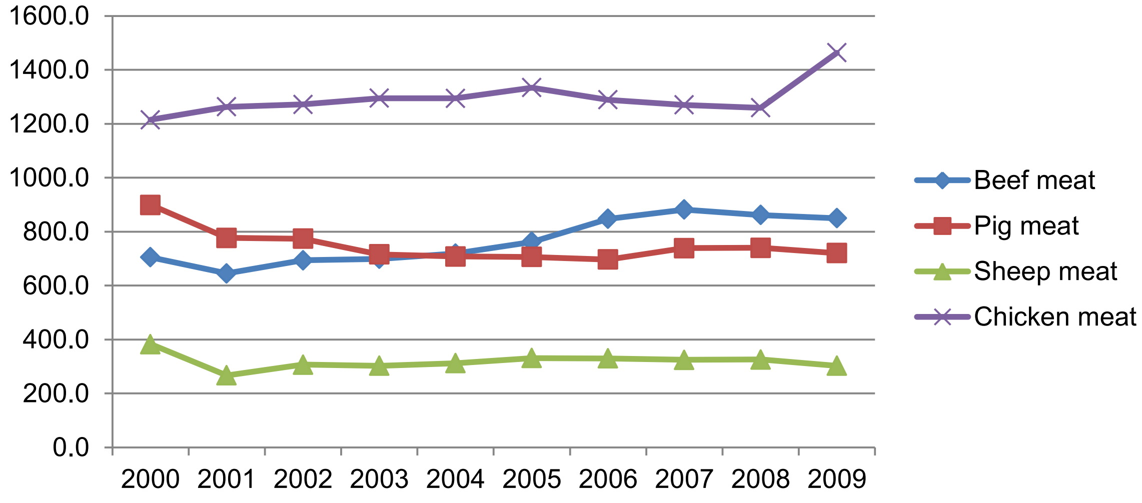

Figure 3 Beef, lamb, pork and poultry meat production in UK in thousand tonnes between 2000 and 2009 (FAO, 2011).

2.1.2 Land based animal production

Beef cattle production has three phases: (i) the cow-calf phase that involves the first 6 to 10 months of a calf's life, during which the animals' weight increases to between 180 kg and 300 kg, (ii) the stocker-yearling phase during which the calf's weight is increased to between 270 kg and 390 kg, mainly by feeding the animal on roughage, and (iii) the feedlot operations phase during which the animal achieves a slaughtering weight (400 kg – 600 kg) at between 15 and 24 months of age. This phase involves high-energy ratio feeds (Taylor and Field, 2009). Dairy cows that are no longer usable for milk production are also sent for slaughter for meat production. In addition, male calves that are born from dairy herds are sometimes brought into the beef system. Consequently, beef production can come from either beef or dairy cattle systems. World cattle stock between 2000 and 2009 was approximately 1,345 million head per annum. Cattle stock in the UK between 2000 and 2009 was variable, there was a downwards trend in stock levels from 11.1 to 9.9 million head (a 11% decrease) (FAO, 2011).

Lambs are raised on pastures and generally do not need feedlot operations. Normally lambs are ready for slaughter at 90 – 120 days of age when they weigh between 40 kg and 70 kg. Only lambs that are not large enough for slaughtering go to feedlots (Taylor and Field, 2009). World sheep stock between 2000 and 2009 was approximately 1,068 million head per annum. The UK has seen a decrease in the sheep stock between 2000 and 2009 from 42.2 to 32.0 million head (a 24% decrease) (FAO, 2011).

Essentially pig production comprises four basic types of operation: (i) Feeder pig production that involves keeping a breeding herd to produce feeder pigs that are kept until they reach a weight of approximately 18 kg, (ii) Feeder pig finishing that involves raising feeder pigs until they reach slaughtering weight (approximately 100 kg), (iii) Farrow-to-finish that involves keeping a breeding herd to produce pigs that are raised until they achieve slaughtering weight, and (iv) Purebred or seedstock that is similar to farrow-to-finish but produces animals for breeding (Taylor and Field, 2009). World pig stock between 2000 and 2009 was approximately 910 million head. During the same period, pig stock in the UK decreased from approximately 6.5 million head to approximately 4.6 million head (a 29% decrease) (FAO, 2011).

Poultry production encompasses the husbandry of chickens, turkeys, geese, ducks, pigeons, peafowls and guineas. Chickens are by far produced in the greatest number. Chicken production includes the incubation of eggs. Young poultry should have adequate feeding and water during the first 10 to 20 weeks, after which, birds are raised and fed until they reach slaughtering weight in 14 – 20 weeks (Taylor and Field, 2009). World chicken stock increased from approximately 14,399 million head in 2000 to 18,631 million head in 2009 (a 29% increase). UK chicken stock increased by 3% in the same period, from 154.3 to 159.3 million head chicken (FAO, 2011).

2.1.3 Slaughtering and meat processing

The main processes in an animal slaughtering/meat processing plant are: (i) stunning-and-slaughtering, and (ii) dressing the carcass. The terminology used for these processes varies depending on species (for example in the poultry meat processing industry, these are commonly referred as evisceration and portioning, respectively).

Stunning can be performed by mechanical, electrical and gaseous methods. Mechanical methods include the use of penetrating stunners, non-penetrating stunners or bullet guns. In electrical methods an electric current is passed through the animal's brain. Gaseous methods include the use of carbon dioxide (CO_2), whereby the animals are put in chambers containing a mixture of 70% CO_2 and 30% air (Gracey, 1998).

Immediately after stunning, sticking and bleeding is performed that causes death by loss of blood and thus lack of oxygen in the brain. This can be accomplished by cutting the throat or thoracic inlet (Gracey, 1998).

In the slaughterhouse, depending on the species, animals are divided mainly into lean carcass, skins and hides, and slaughterhouse by-products. Dressing of the lean carcass differs depending on the animal species. The basic processes include bleeding, skinning, removal of offal and carcass splitting (Gracey, 1998).

On average the carcass weight of cattle, sheep and pigs accounts for 53%, 47% and 75% of the live weight, respectively. The best and worst cases for carcass dressing in the UK for cattle, sheep and pigs are presented in Table 1. Materials that are sent to rendering are defined in the table as "rendering" and specified risk material (SRM). In the best cases defined by EBLEX (2006), rendering material mass is considerably lower than in the worst case for each of the three species (edible material is maximised in the best case). The amount of material sent to rendering can be minimised by good separation practices.

Table 1 Best and worst case of the distribution of final destination of different parts of cattle, sheep and pigs after slaughtering in UK (EBLEX, 2006)

	Cattle			Sheep			Pig		
	kg / head	% of live	cat.	kg / head	% of live	cat.	kg / head	% of live	cat.
carcass wt	318	53%		20	47%		76	75%	
total live weight	599.98			42.14			101.35		
Best case									
carcass lean	192.54	32%		11.7	28%		41.08	41%	
edible human	122.61	20%	3	9.11	22%	3	33.86	33%	3
pet food	4.91	1%	3	1.3	3%	3	1.37	1%	3
rendering	64.46	11%	3	8.5	20%	3	14.89	15%	3
SRM	98.52	16%	1	1.76	4%	1			
hide and skin	42.49	7%	3	4.66	11%	3		0%	3
gut content	74.49	12%	2	5.11	12%	2	10.15	10%	2
Worst case									
carcass lean	192.54	32%		11.7	28%		41.08	41%	
edible human	65.76	11%	3	4.67	11%	3	15.25	15%	3
pet food	0	0%	3	0	0%	3	0	0%	3
rendering	198.43	33%	3	19.35	46%	3	45.02	44%	3
SRM	100.76	17%	1	1.76	4%	1			
hide and skin	42.49	7%	3	4.66	11%	3		0%	3
gut content	0	0%	2	0	0%	2	0	0%	2

The ratio of total materials sent to rendering to carcass lean is in the best case: 0.84 for cattle, 0.88 for sheep and 0.36 for pigs. In the worst case, these ratios are 1.55 for cattle, 1.80 for sheep and 1.10 for pigs. This means in the worst case defined by EBLEX more material is sent to rendering than consumed by humans, whilst in the best case the opposite occurs.

2.1.4 Animal by-products classification and management alternatives

A by-product is a secondary product obtained in the manufacturing of a principal product. Animal by-products (ABP) are secondary products of the animal husbandry and meat industries. ABP includes hides, skins, hairs, feathers, hoofs, horns, feet, heads, bones, toe nails, blood, organs, glands, intestines, muscle and fat tissues, shells and whole carcasses (Meeker and Hamilton, 2006). They are classified into three categories according to European legislation (Table 2).

Table 2 Classification of animal by-products (EC, 2002; 2009)

Category	Material
1	- Animals or body parts with or suspected to be infected with a TSE.
	- Specified risk material (SRM): in the EU: skull, brain, tonsils, spinal cord, and intestines of bovine animals, in UK and Portugal also: entire head and vertebral column.
	- Entire bodies containing SRM.
	- Catering waste from international transport.
	- Animal materials in wastewater from category 1 rendering plants.
2	- Manure and digestive tract content.
	- Animal materials in waste water from slaughterhouses and category 2 rendering plants.
	- Fallen stock (including parts).
	- Not category 1 material.
3	- Parts of slaughtered animals fit for human consumption or unfit but with no transmissible diseases.
	- Hides, skins, horns, bristles and feathers from animal fit for human consumption.
	- Non-ruminant blood.
	- Raw milk from healthy animals.
	- Food of animal origin which is no longer able to be eaten by humans.
	- ABP generated in the production of food products for humans.

The classification was established to reduce animal and human health risks. One of the main drivers of the legislation was the Bovine Spongiform Encephalopathy (BSE) epidemics. BSE is a Transmissible Spongiform Encephalopathy (TSE) that was identified for the first time in the UK in 1986. BSE was described as a pathology that includes the appearance of fibrils and vacuolation in brain grey matter. Initially, cattle with BSE become apprehensive, hyperaesthetic and uncoordinated to walk, ultimately falling (Prince *et al.*, 2003). BSE is of importance because of its association with the rare mortal variant Creutzfeld-Jacob Disease (vCJD), a human TSE. Eating BSE infected cattle may be related to the development of vCJD in humans (Fishbein, 1998).

Methods of treatment of ABP and its final uses or disposal according to their categories are also established in the European legislation (Table 3).

Table 3 Permitted treatments and uses after treatment of ABP (EC, 2002; 2009)

Category	Possible treatments	Permitted uses after treatment
1	Incineration	n.a.
	Rendering	Incineration
		Co-incineration
2	Incineration	n.a.
	Rendering	Incineration
		Co-incineration
		Rendered fats – be used to produce organic fertilizers or oleochemical splitting for technical uses not to be used in humans.
		Rendered proteins – be used as organic fertilizers
		Transformed to produce biogas, composted or landfilled
	Production of biogas	Burn to produce energy
3	Incineration	n.a.
	Rendering	Incineration
		Co-incineration
		Pet foods
		Technical uses
	Production of biogas	Burn to produce energy
	Composting	Compost
	Processing into pet foods	Pet foods
	Processing in a technical plant	Technical uses

Category 1 materials are required to be disposed of by combustion directly or after rendering; however their energy content can be recovered and therefore they can be used as biofuels. Category 2 materials can also be used in composting, biogas production, or after rendering they can be used in fertiliser manufacturing and in the oleochemical industry. Category 3 materials can be used in pet food manufacturing and, subject to previous approval and depending on the species, in the production of animal feeds. The pet food industry can use either chilled or rendered ABP.

A brief description of each option in Table 3 apart from rendering is presented below:

- Incineration: defined as the thermal treatment of material with or without energy recovery (EC, 2000) Incineration provides high security in the destruction of organic material (Woodgate, 2006).

- Co-incineration: defined as the burning of the material in an energy or material generation plant (EC, 2000).

- Anaerobic digestion (AD): defined as the decomposition of the ABP by microorganisms in the absence of oxygen resulting in the production of biogas, which can be burnt to produce heat and power, or used as a road transport fuel. For ABP this in practice typically requires the addition of high carbon materials such as straw (Woodgate, 2006).

- Composting: defined as the aerobic decomposition of the ABP by microorganisms in the presence of oxygen. Typically ABP are in practice co-composted with high carbon materials (Woodgate, 2006). Co-composting has been found to be an effective way of disposing of cattle mortalities (Xu *et al.*, 2007). Furthermore the practical viability of co-composting as a way of disposal of SRM has been investigated with success (Hao *et al.*, 2009). The latter is not permitted in the EU.

- Preparation of pet food: includes the use of fresh or frozen ABP. The amount of pet food that uses chilled ABP is decreasing globally, while the amount of dried pet food which uses rendered products as ingredients is increasing (Woodgate, 2006).

- Burial/landfill: this option is not permitted in the EU for raw ABP; however globally it is considered an option. This option is associated with high risks of spreading animal diseases (Woodgate, 2006).

In general, different variations of incineration/co-incineration, rendering, composting and generation of biogas in vessels are perceived as preferred options for the safe disposal of carcass material. Landfill and composting in windrows are considered secondary options. Anaerobic digestion in pits and on-farm burial are considered undesirable options (Pollard *et al.*, 2008).

2.1.5 Rendering

Woodgate and van der Veen (2004) define rendering as "to render open or split - by heat processing – raw material into a solid (protein meal also referred to as Meat-and-Bone Meal: MBM) and a liquid (fat in the form of tallow is a liquid at elevated temperatures)". In practice the term has become related to the processing of inedible ABPs. There are two main streams of ABP: fallen stock (on farm) and slaughterhouse/meat processing by-products (Figure 1).

Fallen stock from animal agriculture represents between 3% and 10% of the total throughput of the rendering industry (Bansback, 2006). In the UK, the most common methods of fallen stock disposal used by farmers are: Knackers' yards, hunt kennels, incineration and rendering (DEFRA, 2008; Kirby *et al.*, 2010). The first two seem to be the preferred options. It should be noticed, that Knackers' yards are intermediaries, and thus the final destination of the farm ABP that go to this disposal option might be rendering. The main stream of ABP comes from the slaughterhouses/meat plants.

2.1.5.1 The rendering process

There are two main rendering systems: dry and wet systems (Woodgate and van der Veen, 2004).

Dry rendering is the prevalent system currently (Anderson, 2006). In the dry continuous process the material is ground and passed through a disc dryer/cooker. The vapour from the cooker is taken to a condenser. Process condensate is typically sent to wastewater treatment and non-condensable gases are treated to destroy odours. The cooked material is directed to a press system where the liquid (fat) is separated from the solid (proteinaceous material). The solid is taken typically to grinding and screening systems to obtain protein meal. The liquid is directed to a filtration system to obtain rendered fat (tallow in the case of beef material). Figure 4 presents a diagram of a continuous dry process.

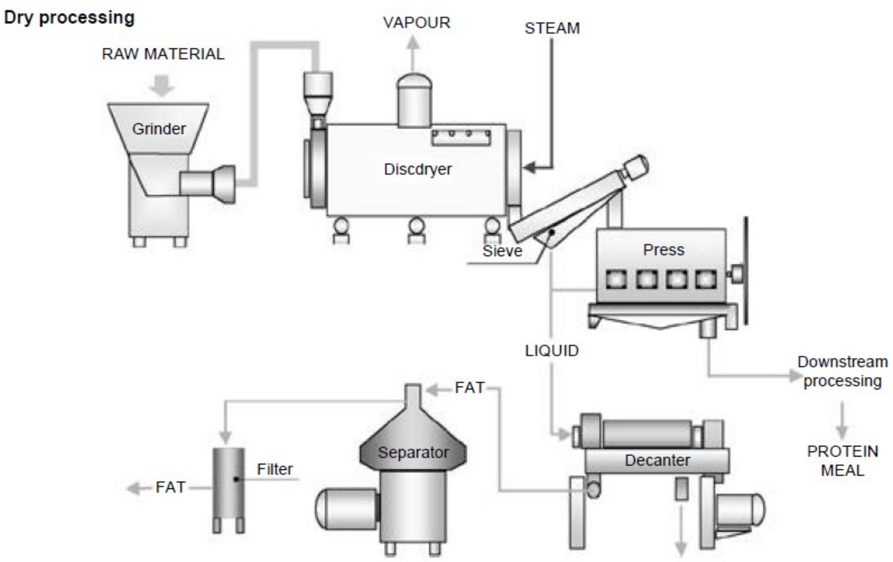

Figure 4 Dry rendering process (Woodgate and van der Veen, 2004)

Similarly, in the wet process the heat provided melts the fat; however both meal and fats remain hydrated until a later stage. Further evaporation takes place in another step (Woodgate and van der Veen, 2004). Currently, wet rendering is more associated with edible rendering (Anderson, 2006). Figure 5 presents a diagram of a wet processing system.

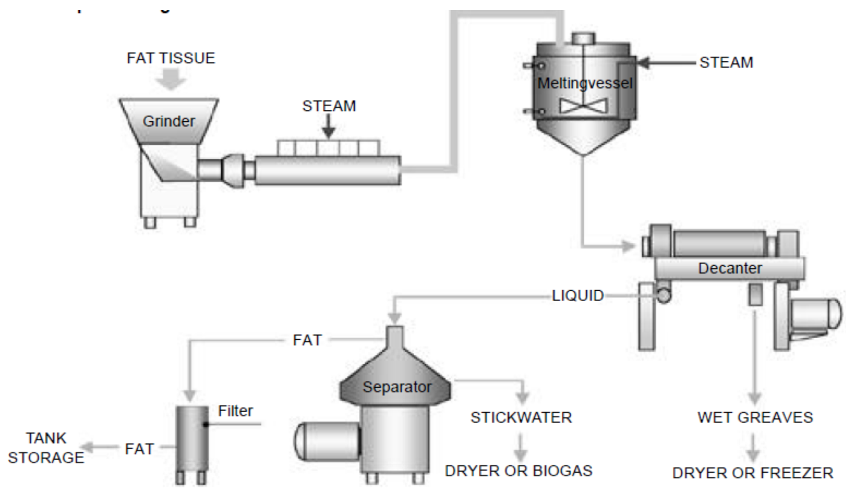

Figure 5 Wet rendering process (Woodgate and van der Veen, 2004)

Kalbasi-Ashtari et al. (2008) stated that the chemical composition of raw materials, particle size, type of rendering system, air pressure and temperature influence the cooking time required. High pressure and temperature are associated with a reduction in the potential BSE infectivity in the protein meal; however it also decreases its nutritional value since it affects some amino acid content and digestibility. The particle size of raw material is very important because smaller particles lead to greater heat penetration.

In the EU legislation, 5 methods for rendering processes (not including fish rendering) are approved. Table 4 presents the different methods, which mainly define the maximum particle size before entering the cooking process and the time, temperature and in some cases pressure required during the cooking process.

Table 4 Raw material size, temperature and pressure for different methods (EC, 2002; 2009)

Method	Maximum Raw Material particle size [mm]	System	Cooking parameters[a]: Core Temperature (T) [°C], time (t) [min], and Pressure (P) [bar]
1	50	Batch or continuous	T = 100, t = 20, P = 3
2	150	Batch	1st step: T = 100, t = 150
			2nd step: T = 110, t = 120
			3rd step: T = 120, t = 50
3	30	Batch or continuous	1st step: T = 100, t = 95
			2nd step: T = 110, t = 55
			3rd step: T = 120, t = 13
4	30	Batch or continuous	1st step: T = 110, t = 13
			2nd step: T = 120, t = 8
			3rd step: T = 130, t = 3
5	20	Batch or continuous	Heating until coagulation, and then pressed to remove fat and water
			Proteinaceous material:
			1st step: T = 80, t = 120
			2nd step: T = 100, t = 60

[a]When the cooking parameters include more than one step, the time-temperature requirements can be achieved simultaneously.

It has been suggested that BSE transmission is associated with the intake of rendered products included in cattle feeds (Fishbein, 1998; Prince *et al.*, 2003). Furthermore, it has been suggested that changes that took place in rendering technology in the early 1980's caused BSE to survive the rendering process: in particular the change from batch to continuous rendering and the removal of the solvent extraction phase. Solvent extraction involved the use of an organic solvent at high temperatures to extract the protein meal, which was then exposed to high temperature steam to remove the solvent. Additionally at that time the rendered protein meal fraction in feeds was increased from 1 % to 12 % (Fishbein, 1998). Regarding rendered fats, the number of publications associated with the significance of tallow for BSE transmission are limited (Woodgate and van der Veen, 2004).

The BSE transmission issue was unknown until the studies on the epidemics. Taylor *et al.* (1995) analysed 15 rendering methods, 12 of which were in use in the EU at the time of the epidemics to verify their effectiveness for inactivation of the BSE agent. The tests included BSE infected raw material. The results indicated that 4 processes produced MBM with BSE infectivity. No infectivity was found in tallow produced by the 15 processes. Processes that allowed survival of the BSE agent are no longer used in the rendering industry.

2.1.5.2 Animal by-products processed by rendering in United Kingdom

There are 22 rendering plant locations in the UK according to the UK Rendering Association (UKRA - Figure 6). Plants are located across the whole UK. They are normally located outside highly populated areas, however there are some plants located in urban areas.

In 2006, 2007, and 2008 the rendering industry in the UK processed around 2 million tonnes of ABP per year (Figure 7). Total UK meat production in 2006, 2007 and 2008 was 3.39, 3.41, and 3.36 million tonnes respectively (FAO, 2011). Consequently ratios of ABP processed by rendering to meat produced were 0.61, 0.65 and 0.60 in 2006, 2007 and 2008, respectively. This means the amount of material rendered was between 60% and 65% of the total meat produced in the UK (assuming that the fallen stock amount is not important). Thus the amount of biomass processed by the rendering industry can be considered significant (in the same order of magnitude as that of total meat production). These ratios are consistent with the carcass to rendering material ratios for cattle, sheep and pigs presented in section 2.1.3. Figure 7 presents the amount of ABP treated by rendering according to the ABP categories; categories 1 and 2 are quantified together because they are normally handled together as category 1 material. Between 2006 and 2008 approximately 60% of the ABP treated was category 1 and 2 material, whilst in 2008 approximately 57% of the ABP treated was category 3 material.

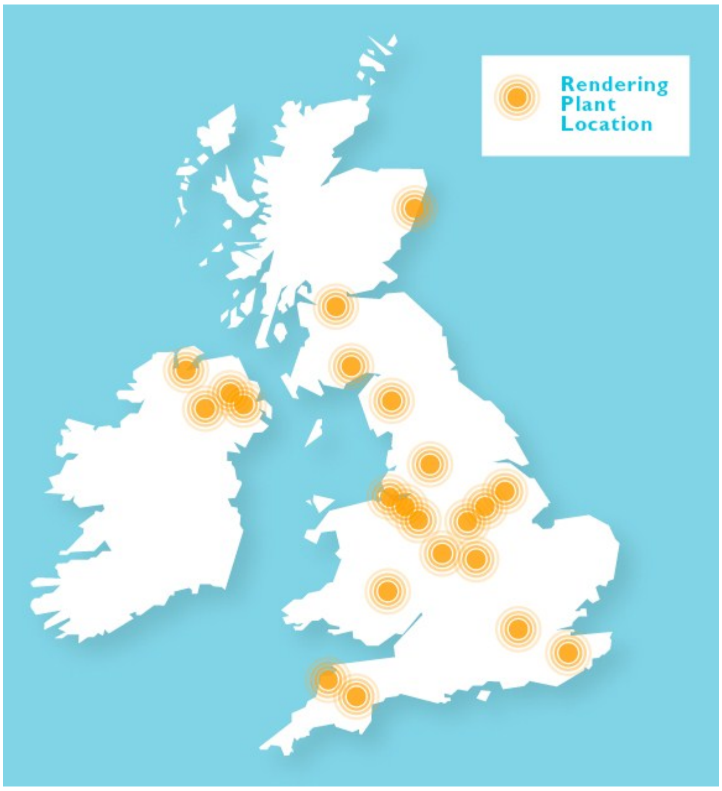

Figure 6 Rendering plants location in UK (UKRA, Not dated)

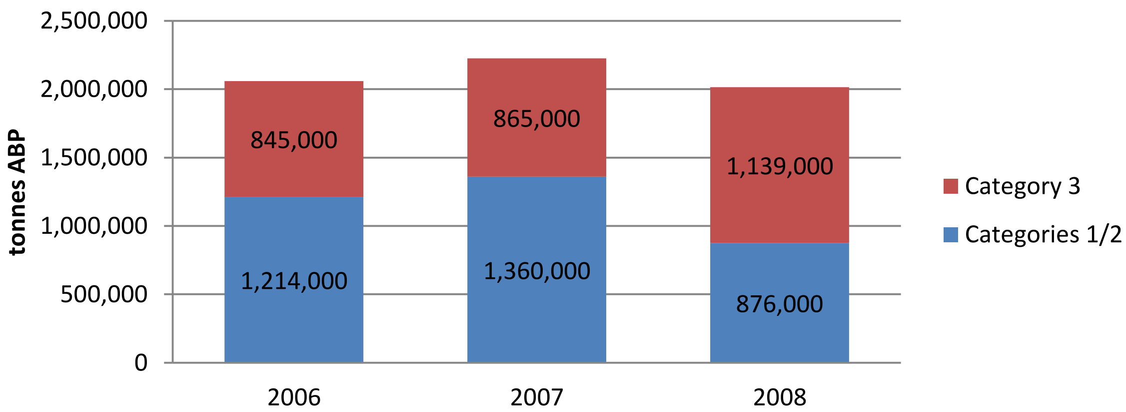

Figure 7 Total animal by-products amount handled by category in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

ABPs can also be classified by species, as shown in Figure 8. SRM is always ruminant material (cattle or lamb). Mixed material is difficult to define, but includes fallen stock and slaughterhouse by-products as well as supermarket/food waste. In 2008 the percentage of SRM was very high in comparison to years 2006 and 2007. The SRM percentage was approximately 15% in 2006 and 2007 and 35% in 2008. The mixed material portion was lower in 2008; thus maybe the increase in SRM is related to more detailed statistics in 2008.

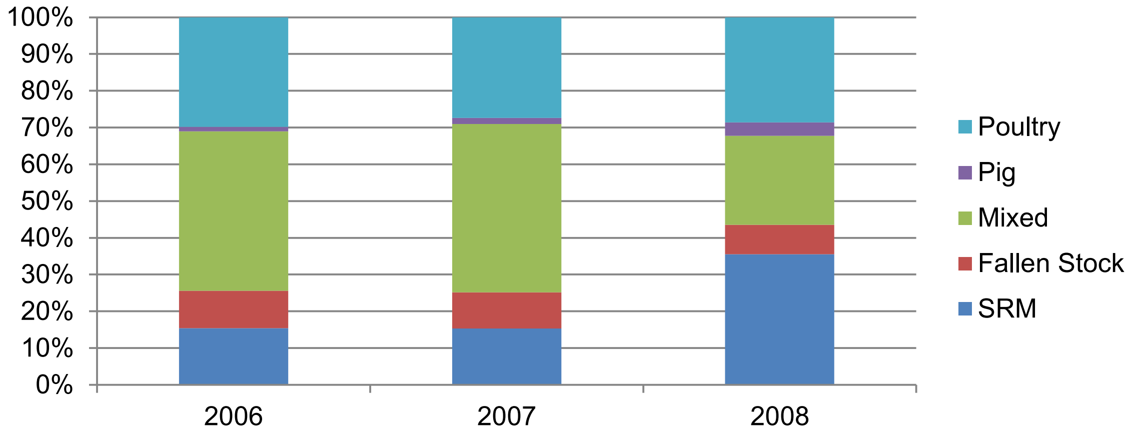

Figure 8 Distribution of animal by-products amount by species in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

2.1.5.3 Rendered products and their uses in United Kingdom

The products of the rendering industry are rendered proteins and fats. The total production of dry rendered products (DRP) in the UK for years 2006, 2007, and 2008 is presented in Figure 9. Over this period there was an increase in the production of DRP. However, the processing of ABP has been stable or even decreased. Of the total DRP produced, proteins represented 64%, 65% and 61% for years 2006, 2007, and 2008 respectively. The yield of DRP (ratio of DRP produced to ABP processed), was 35%, 34% and 41% respectively.

Figure 10 presents the tonnage of DRP produced by category and by type (fat or proteins) for the years 2006, 2007 and 2008. As with ABP category 1 and 2 products are combined. Category 1 and 2 products account for more material than category 3 products. Category

1 rendered proteins are still known as MBM, whilst current legislation (EC, 2009), defines category 3 proteins as Processed Animal Proteins (PAP).

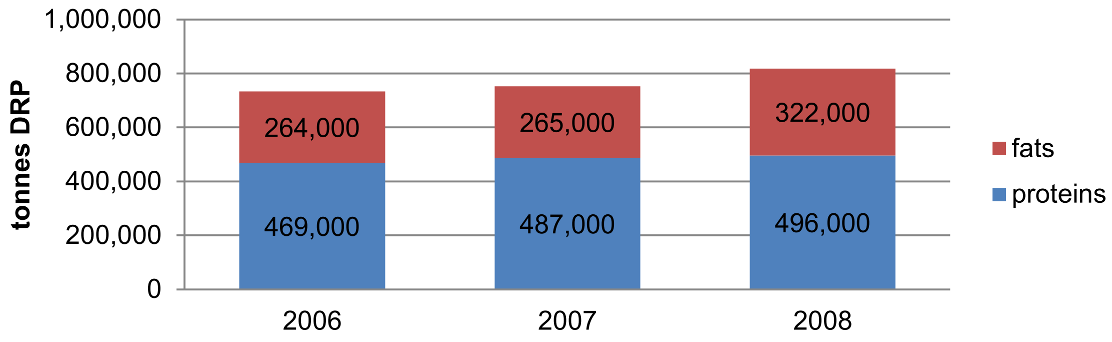

Figure 9 Production of DRP in UK for years 2006, 2007 and (Pers. Comm. SL Woodgate 2010)

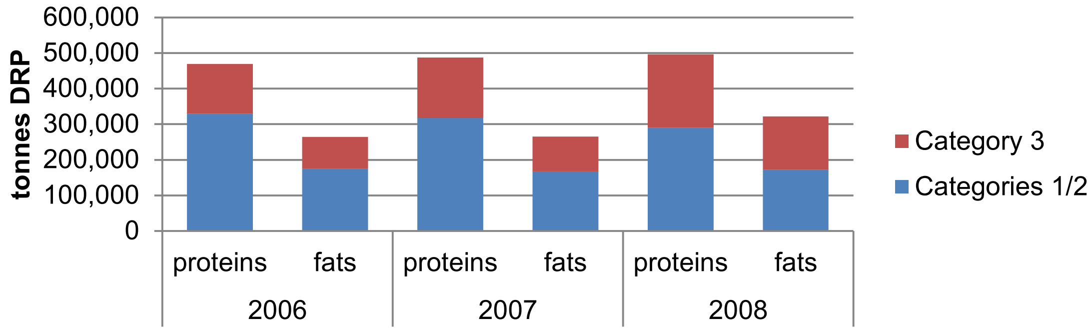

Figure 10 Production of DRP by categories in UK for years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Table 5 presents the different uses of rendered proteins in 2006, 2007 and 2008. Most proteins are used as fuels and in the pet food industry.

Table 5 Use of rendered proteins in tonnes by category for years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Use	Category	2006	2007	2008
Combustion fuel	1/2	344,000	262,000	213,500
Cement	1/2	0	55,000	83,500
Cement	3	0	5,000	0
Animal Feed	3	14,500	10,000	13,000
Pet Food	3	97,500	127,000	170,000
Fertiliser	3	13,000	28,000	16,000

Category 3 proteins are used mostly in the pet food industry (Fig 11). Other uses of category 3 proteins include animal feed and fertilisers. Despite the fact category 3 products can be used in these valuable applications, in 2007 small amounts were used in the cement industry as fuel in kilns.

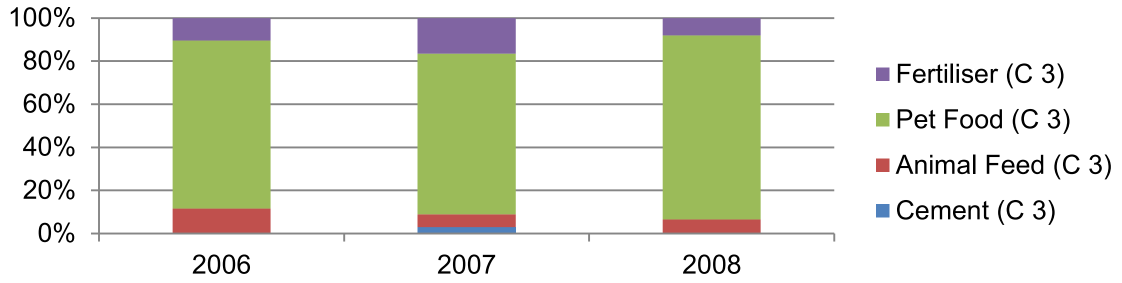

Figure 11 Use distribution of category 3 rendered proteins in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Category 1 and 2 rendered proteins are treated as category 1 materials. Therefore they both are used as fuels in the cement industry for heat production or in fluidised bed combustion (FBC) plants for the production of heat and/or power. The latter was the most prevalent use in 2006, 2007 and 2008 (Figure 12).

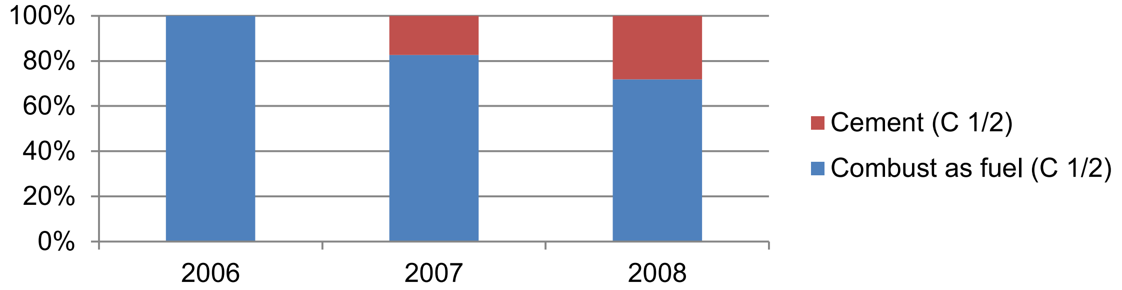

Figure 12 Use distribution of categories 1 and 2 rendered proteins in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Table 6 presents the different uses for rendered fats in years 2006, 2007 and 2008. In the case of fats, some uses are not separable by categories, as their use depends on the market price. Combustion fuel refers to the use of rendered fats as fuels in boilers for heat production in the rendering industry itself. Only category 1 and 2 materials are used as fuels in the rendering industry.

Table 6 Use of rendered fats in tonnes by category for years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Use	Category	2006	2007	2008
Combustion fuel	1/2	118,000	130,000	98,000
Cement fuel	1/2			67,000
Biodiesel	1/2/3	20,000	47,500	40,000
Oleochemicals	2/3	49,000	76,500	69,000
Pet Food	3	21,000	11,000	34,000
Animal feed	3	26,000		14,000
Store	1/2	30,000		

Figure 13 presents the percentage use of category 2 and 3 fats for non-combustion purposes. The main use of these products is as raw materials in the oleochemical industry. Category 3 fats are also used in the manufacturing of pet foods.

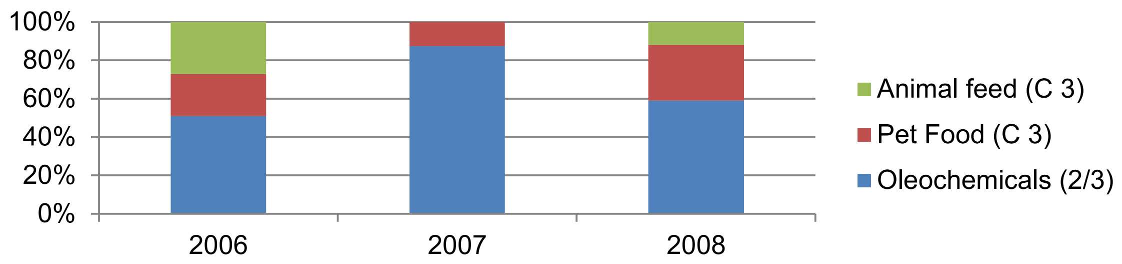

Figure 13 Use distribution of category 2/3 rendered fats in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

Figure 14 presents the percentage use of fats for combustion purposes. Their use as fuel replacement in the rendering industry itself is the most important. In 2008, their use as fuels for the cement industry was also important.

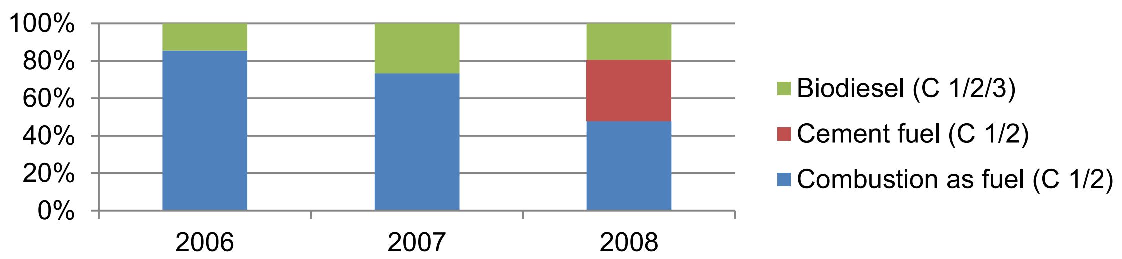

Figure 14 Use distribution of category 1/2/3 rendered fats used for combustion in different systems in UK in years 2006, 2007 and 2008 (Pers. Comm. SL Woodgate 2010)

2.2 Application of life cycle assessment methodology

2.2.1 Sustainable development

The term sustainable development has become very important in the last decades; it is defined in the Brundtlandt Report (UN, 1987) as "development that meets the needs of the present without compromising the ability of future generations to meet their own needs. It contains two key concepts:

- the concept of 'needs', in particular the essential needs of the world's poor, to which overriding priority should be given; and
- the idea of limitations imposed by the state of technology and social organization on the environment's ability to meet present and future needs."

This definition can be perceived as quite broad, and efforts towards a more specific definition have been made. Jabareen (2008) has described a conceptual framework for sustainable development (Figure 15).

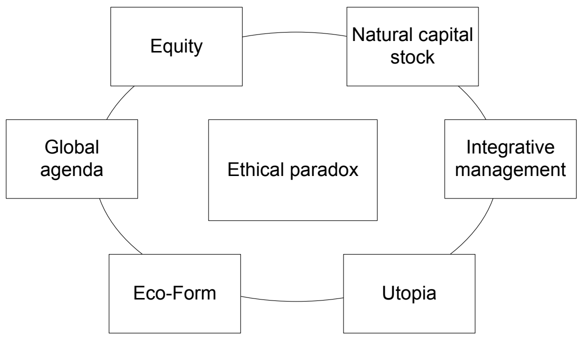

Figure 15 Conceptual framework for sustainable development (Jabareen, 2008)

The seven concepts in the Figure 15 are useful to illustrate the implications and requirements of sustainable development. In the core, the ethical paradox refers to the actual words; on one hand "sustainability" is associated with a state that can be kept indefinitely and on the other hand, "development" requires modification. The natural capital are natural assets that humans can modify but not create, and is normally divided into three categories: non-renewable resources, the capacity of nature to produce renewable resources, and the capacity of nature to assimilate pollutants. An associated term is "strong sustainability" which implies that the natural capital should be kept constant. In contrast, "weak sustainability" allows for the intervention in the natural system (OECD, Not dated).

Equity is related with the social aspects of sustainable development. It is associated with environmental, social and economic justice, social equity, equal rights, quality of life, right distribution, freedom, democracy, public participation and empowerment. There are two types of equity: intergenerational and intragenerational. Intergenerational is related to equal distribution between present and future generations. Intragenerational is associated with the fairness in the distribution of resources at the present time (Jabareen, 2008).

The Eco-form is a term associated with the design of urban places, buildings and houses. Typical associated technologies and ideas are: alternative materials, renewable energy, organic food, conservation and recycling. Energy efficiency is seen as one of the core elements in this sense. The concept of utopia is associated with sustainable development and a vision of a perfect society, with justice and in harmony with nature. This vision would require deep changes towards a non-competitive and non-materialistic society (Jabareen, 2008).

Integrative management refers to the integrative and holistic approach required for sustainable development. Social development, economic growth and environmental protection are to be considered together. The political global agenda refers to the shift towards a global agenda in contrast to traditional local and national environmental protection schemes. The global political agenda is also attached to the integrative approach in the sense that sustainable development should deal with world poverty eradication, shift in consumption and production and adequate care of the natural system in contrast to the traditional only environmental approach (Jabareen, 2008).

The concept of sustainable development is a very broad one and seems to be associated with many issues. Environmental, social and economic dimensions are the three dimensions of sustainability. However, some priorities can be made. Environmental sustainability also in itself has different aspects. In the current political agenda, one of the most important environmental issues is climate change and mitigation of greenhouse gas emissions. To make these concepts practical, there is the need for tools that allow comprehensive assessment of the environmental burden and resource use from different production systems. These tools should be flexible enough to focus on particular environmental criteria, for example, climate change.

2.2.2 Climate change

"Climate change refers to a change in the state of the climate that can be identified (e.g., by using statistical tests) by changes in the mean and/or the variability of its properties, and that persists for an extended period, typically decades or longer. Climate change may be due to natural internal processes or external forcings, or to persistent anthropogenic changes in the composition of the atmosphere or in land use" (Solomon *et al.*, 2007a).

The mean temperature of the Earth depends on the heat from the Sun and the properties of the Earth. To maintain an inhabitable temperature the natural greenhouse effect traps infra-red radiation within the Earth's atmosphere. This is achieved by Greenhouse Gases (GHG's) that absorb a proportion of the infra-red radiation that would otherwise be lost to space. Since the industrial revolution the concentration of GHG's in the atmosphere has increased due to anthropogenic activities resulting in an increment in the atmospheric absorption of outgoing infra-red radiation. There are six main anthropogenic GHGs: carbon dioxide (CO_2), methane (CH_4), nitrous oxide (N_2O), Ozone-depleting substances (ODS), chlorofluorocarbons (CFCs) and hydroclorofluorocarbons (HCFCs). Some of these gases normally exist in the atmosphere, however during the last 250 years the increase has been attributed to human activities (Solomon *et al.*, 2007a).

GHGs can be characterized by the Global Warming Potential (GWP), which is expressed in relation to that of CO_2. GWP for a 100-year time horizon of CO_2 is 1, for CH_4 is 25, and for N_2O is 298 (Solomon *et al.*, 2007a). The time horizon of 100-year is commonly used by regulators and the literature. CH_4 and N_2O are considerably more powerful GHGs than CO_2. However CO_2 is often considered the most important GHG due to its abundance (Figure 16).

Projected concentration increases are used to estimate temperature increases. Temperature increase by the end of the 21st century (2090 – 2099) could be between 1.1 °C and 6.4 °C relative to that of years 1980 – 1990. An associated sea level rise of 0.18 m to 0.58 m in the same period is also projected. Extreme weather situations are predicted to be more frequent (Solomon *et al.*, 2007a). Depending on the temperature increase different impacts can be expected (Table 7). Climate change is a serious threat for humans and nature, and thus represents the most important environmental sustainability challenge for humanity in the 21st century.

Table 7 Potential Impacts by the end of the 21st century associated with different temperature increase above preindustrial era (Great Britain, 2006)

Temperature increase above preindustrial	Potential Impacts
1- 2 ºC	Major impacts on ecosystems and species
2-3 ºC	Greenland ice cap starts to melt, major loss of coral reef ecosystem; considerable species loss; large impacts on agriculture; Terrestrial carbon sink could become a source
1-4 ºC	North Atlantic circulation at increasing risk of collapse
2-4.5 ºC	West Antarctic ice sheet at increasing risk of collapse

In 2004, globally the most important GHG was CO_2 (Figure 16). In particular the combustion of fossil fuels for heat and power (and transport) was the greatest source of CO_2 followed by deforestation and decay of biomass. CH_4 is the second most important GHG; its main source is agriculture. N_2O emissions are also associated with agriculture due to the use of fertilisers (Barker *et al.*, 2007).

The major source of GHGs in the world is energy supply. Other major sources are industry, forestry, agriculture and transport (Figure 17). In energy, transport and industry, the main GHG is CO_2, which is a very different situation for that of agriculture where the main GHG's are methane and nitrous oxide.

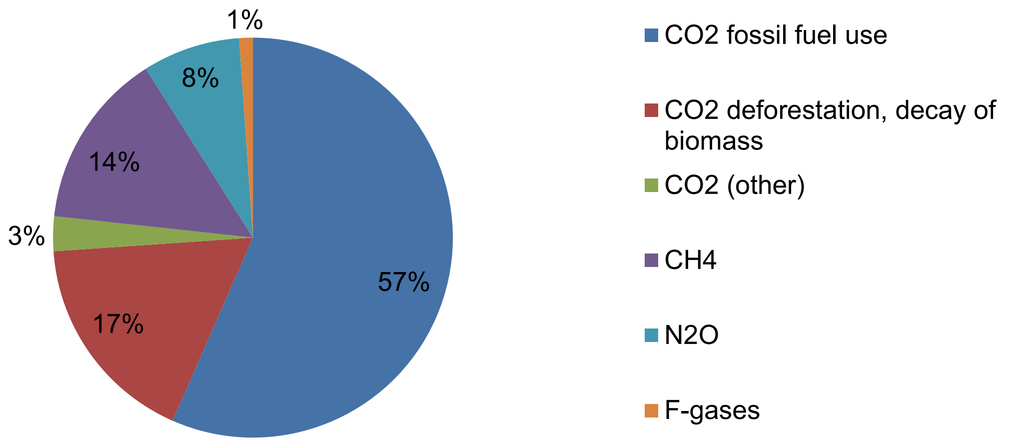

Figure 16 Global anthropogenic GHG emissions in 2004 (Barker *et al.*, 2007)

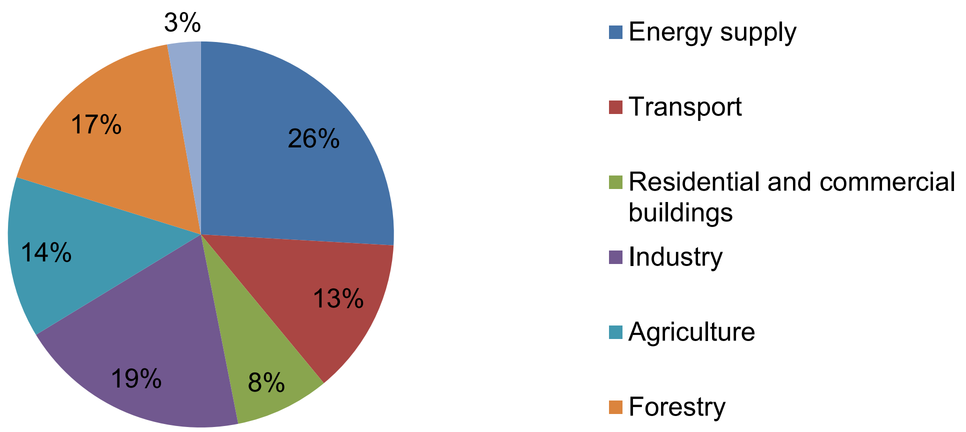

Figure 17 Contribution of global GHG emissions by sector in 2004 (Barker *et al.*, 2007)

Similarly in the UK, the sector with the highest GHG emissions was Energy supply that accounted for 35 % of emissions in 2009 (Figure 18). Transport was the second and business was the third largest contributors with 22% and 15% respectively. GHG emissions from agriculture were 49.5 million tonnes CO_2 equivalents (9% of the total in 2009). Industrial processes accounted for only 2%. The reported figures are based on sectorial emissions (e.g. energy supply is the whole energy supply to the different sectors in the UK). A life cycle perspective (instead of sectorial) would include the emissions associated with energy production in the sector where they are used. There has been a reduction of 28% in the GHG emissions from the UK between 1990 and 2009. The emission reduction in Energy supply, Business, Waste Management and Industrial

Processes accounted together for 87% of this reduction. The Climate Change Act 2008 (UK, 2008a) sets a reduction target of 80% in relation to the level of 1990 for year 2050, which indicates that by 2050 total GHG emissions must be less than 156 million tonnes CO_2.

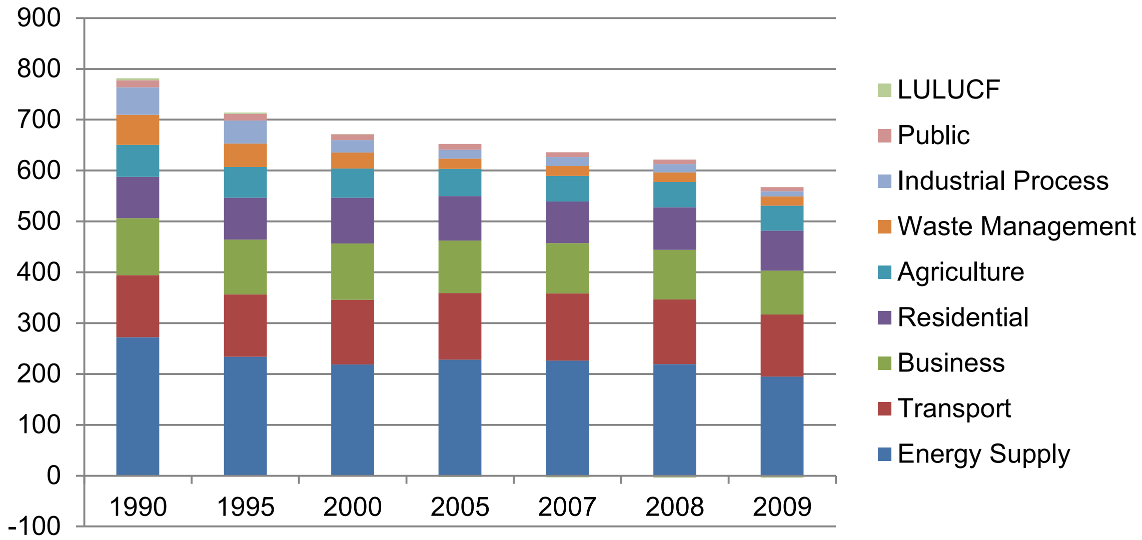

Figure 18 Evolution of GHG emissions in the UK by sector in million tonnes CO_2 between 1990 and 2009 (DECC, 2011) LULUCF: Land Use, Land Use Change and Forestry

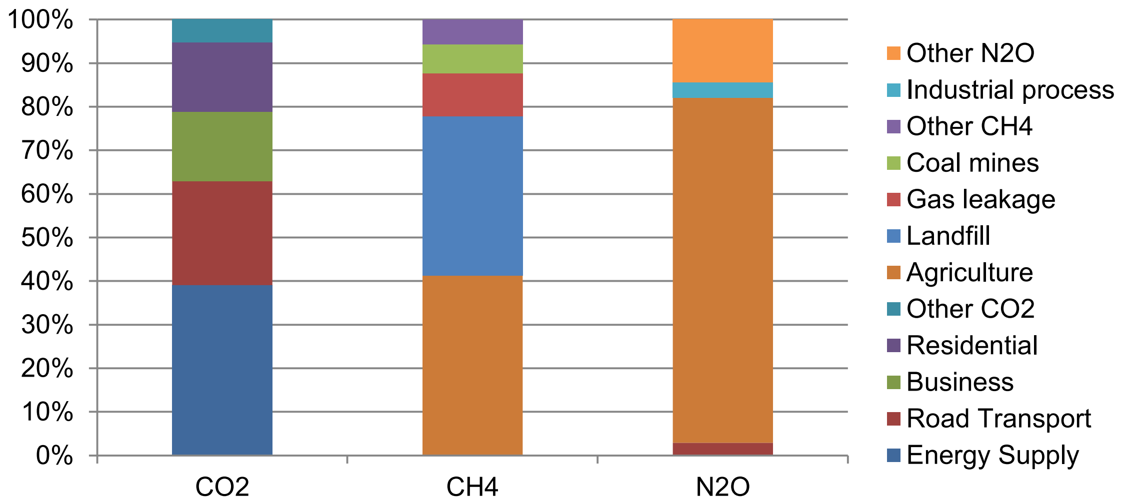

Figure 19 Main contributors to each GHG in the UK (DECC, 2011)

In 2009, the emissions of GHG as CO_2, CH_4, and N_2O accounted for 84%, 8% and 6% of the GHG emissions in the UK, based on CO_2 equivalents (DECC, 2011). The sectorial contribution was different for each GHG, with the major contributors to CO_2 related to those sectors that heavily depend on the combustion of fossil fuels: Energy Supply and Transport. Agriculture is the main contributor to emissions of CH_4 and N_2O, the former mainly associated with anaerobic decomposition (in enteric fermentation and manure management) and the latter mainly associated with fertiliser application and manure management. Anaerobic decomposition in landfill is also an important source of CH_4 (Figure 19).

There are two types of strategies that are addressed regarding climate change and the future: mitigation and adaptation. Mitigation refers to "a human intervention to reduce the sources or enhance the sinks of greenhouse gases" and adaptation to "Initiatives and measures to reduce the vulnerability of natural and human systems against actual or expected climate change effects" (Solomon *et al.*, 2007a).

The shift towards more efficient and renewable energy systems is seen as one of the most important mitigation activities. This involves the shift from fossil fuel systems (coal, oil and gas) to carbon neutral or less carbon intensive energy systems (hydropower, wind, bioenergy, geothermal, solar, nuclear, etc.). It is important to realise that carbon neutrality (no CO_2 emissions) does not necessarily imply total climate friendliness as there are other powerful GHG's as N_2O and CH_4. Comprehensive assessments of systems should be undertaken to find holistic solutions.

2.2.3 Sustainability assessment tools

The evaluation of sustainability requires methods to measure the current status or effects of changes. A great variety of sustainability assessment tools have been developed. Ness *et al.* (2007) have classified them in three areas of coverage: indicators/indices, integrated assessment and product-related assessment.

Indicator and indices are measures that are associated to some state of social, economic or environmental development in a region (Ness *et al.*, 2007). Indices are aggregated indicators. Examples of indicators are the United Nations Indicators of Sustainable Development (UN, 2009). These are indicators at national level related to social (e.g. Gini

Index of Income Inequality), economic (e.g. Gross Domestic Product per capita), environmental (e.g. Emission of GHG) and institutional (e.g. Number of Internet Subscribers per 1000 Inhabitants) issues. Integrated indexes are oriented towards the integration of nature and societal dimensions in one index, examples are the Human Development Index by the UN Development Programme, which integrates life expectancy, literacy and standard of living (Ness *et al.*, 2007).

Integrated assessment refers to tools that are used in the support of decision making related to projects and policies (Ness *et al.*, 2007). One example of this type of tool is Environmental Impact Assessment for projects.

Product-related assessments are associated with flows in production and consumption of goods and services. Of these, Life Cycle Assessment (LCA) is the most established tool, and has been used to evaluate environmental aspects in different product and service systems (Ness *et al.*, 2007). The basis for assessment based on products (and services) is that the environmental impact of the economy is related to the consumption of products (and services), directly by the actual use of them and indirectly by their production and final disposal (Tukker and Jansen, 2006).

LCA is a tool that was developed for the evaluation of industrial products; however it has been successfully used to evaluate the environmental impact of agricultural products (Williams *et al.*, 2006; de Vries and de Boer, 2010). Rendered product chains involve both agriculture and industrial processes. The suitability of LCA to be used in products from different sectors like industry, agriculture and energy, makes it an adequate tool to evaluate the energy use and GHG emissions from rendered product systems.

2.2.4 Life cycle assessment

LCA is a tool for the assessment of the environmental impact of a product or service throughout every stage of its life cycle. LCA methodology is standardized by international standards ISO 14040 (ISO, 2006a) and ISO 14044 (ISO, 2006b). LCA is defined in the ISO standards as the “compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle”. The life cycle is the “consecutive and interlinked stages of a product system, from raw material acquisition or generation from natural resources to final disposal” and a product system is the “collection of unit processes with elementary and product flows, performing one or more defined functions, and which models the life cycle of a product” (ISO, 2006a).

LCA involves four phases: goal and scope definition, inventory analysis, impact assessment, and interpretation. These phases have a logical order, however LCA has an iterative character and anything can be refined in any phase of a study.

2.2.4.1 Goal and scope definition

The objectives of the study are defined in this phase. This involves the justification for the study and the intended use of the results. The scope of the study includes definition of the product system and functional unit. The functional unit is the “quantified performance of a product system for use as a reference unit” (ISO, 2006a). Table 8 presents some examples of functional units for different product systems.

Table 8 Example of functional units

Product system	Functional unit
Beverage packaging	Litres of packaged drink
Beef	kg of beef meat
Painting	m^2 x year

The system boundary is defined to indicate the processes that are part of the product system. Processes to be included depend on the objectives of the study.

The ISO standards present a list of unit processes that can be considered as part of a product system: “acquisition of raw materials; inputs and outputs in the main manufacturing/processing sequence; distribution/transportation; production and use of fuels, electricity and heat; use and maintenance of products; disposal of process wastes and products, recovery of used products (including, reuse, recycling and energy recovery); manufacture of ancillary materials; manufacture, maintenance and decommissioning of capital equipment; additional operations, such as lighting and heating”.

The methodological framework can be used for: cradle-to-gate studies, gate-to-gate studies and even specific parts of a production system (like waste management, components of a product system, etc). Figure 20 illustrates the difference between cradle-to-gate, gate-to-gate and cradle-to-grave studies.

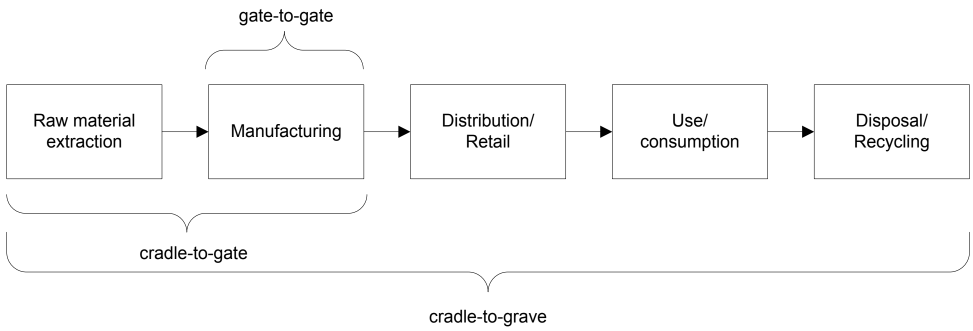

Figure 20 Differences between cradle-to-gate, gate-to-gate and cradle-to-grave

There is a consensus that there are two types of LCA: attributional (called also accounting or retrospective) and consequential (called also change oriented or prospective) (Baumann and Tillman, 2004; Curran, 2007a; Finnveden *et al.*, 2009). The main differences regard boundaries, co-product handling procedures and choice of data. In attributional studies, average data is used while in consequential, marginal data is used (Baumann and Tillman, 2004). Average and marginal data issues are common when studying systems that use or produce electricity, where average represents the national electricity technology/fuel mix (global annual average) and marginal is associated with the technology/fuel that is used on the margin (e.g. used during the peak load of the day). Average and marginal data issues also appear in the case of the production of farmed animal feeds, where average sources of fats and proteins are normally different to marginal sources of fats and proteins (Dalgaard *et al.*, 2008; Schmidt, 2010).

Cut-off criteria should also be established. This helps to identify which inputs are to be included in the study. Mass, energy or environmental significance can be used as cut-off criteria. A contribution percentage should be defined for which the total sum of inputs should account. Mass cut-off criteria would involve the setting of a percentage (e.g. 95%), of the total input mass to the system, for which the sum of the inputs included in the study should be at least equal to 95% (ISO, 2006b).

2.2.4.2 Inventory analysis

In this phase, all the data collection and calculation of inputs and outputs is performed. Qualitative and quantitative data for each unit process in the product system should be collected by measuring, calculation or estimation. A process flow diagram, that includes the processes and their relationships, should be developed. Data collected might be classified as suggested in ISO 14044:

"- energy inputs, raw material inputs, ancillary inputs, other physical inputs,

- products, co-products and waste,

- releases to air, water and soil, and

- other environmental aspects"

The calculation includes the accounting of all the inputs and outputs of processes in the flowchart. A software tool that provides a framework to organise the product system and perform the calculations is typically used (e.g. Simapro (PRe Consultants, 2011)). The use of commercial databases which have access to common products is a general practise in LCA.

The results of an inventory analysis are the product and elementary flows involved in the product system, normalised according to the functional unit. A product flow (economic flow) is "products entering from or leaving to another product system" and an elementary flow (natural flow) is "material or energy entering the system being studied that has been drawn from the environment without previous human transformation, or material or energy leaving the system being studied that is released into the environment without subsequent human transformation" (ISO, 2006b).

2.2.4.2.1 Co-product handling

Co-product handling is required when a process produces more than one co-product. A co-product is "any of two or more products coming from the same unit process or product system" (ISO, 2006a). Allocation is "partitioning the input or output flows of a process or a product system between the products system under study and one or more other product systems".

Wastes are also outputs; however inputs and outputs should be allocated to co-products only. Wastes in LCA are "substances or objects which the holder intends or is required to dispose of" (ISO, 2006b). Econometrica *et al.* (2009) studied the indirect GHG missions from the use of wastes, residues and by-products in the UK. They recognised that the terms "waste", "residues" and "by-products" normally do not have the same definition in legislation and LCA. This can make the use of these terms complex and confusing. ISO 14044 provides a hierarchy regarding co-product handling which suggests that:

- Allocation should be avoided either by process division or system expansion.
- If allocation has to be performed, the inputs and outputs of the system should be divided according to the physical relationships in which inputs and outputs change according to changes in their functions.
- If no physical relationships can be defined, other relationships should be used (e.g. economic value of products and co-products).

Reuse and recycling have special requirements, since these processes (and similar like: composting, energy recovery, etc.) involve more than one product (ISO, 2006b). For these, there are two cases for allocation:

- Closed-loop allocation is used where the properties of the recycled material are the same as the virgin material. Allocation is not needed since the secondary material production reduces the use of the virgin material.
- Open-loop allocation is used when the material is recycled or reused in other product systems.

Allocation and system expansion are among the most debated issues in LCA methodology (Weidema, 1993; Azapagic and Clift, 1999b; a; Ekvall and Finnveden, 2001; Ayer *et al.*, 2007; Curran, 2007a; Reap *et al.*, 2008). However, it is commonly affirmed that there is no single method to solve the multiple output problem (Guinee *et al.*, 2004; Curran, 2007a; Kendall and Chang, 2009). Approaches that are used include:

- Main or primary product (or no allocation): the main product takes all the environmental burden of the system.
- Economic allocation: the economic value ratio of the co-products is used to divide the environmental burden among co-products.
- Mass allocation: the mass proportion of the co-products is used to divide the environmental burden (a similar approach can be taken with volume).
- System expansion: the avoidance of allocation through the inclusion of avoided products from other products systems in the studied system.
- Biological allocation: it has been argued that biological causality could be used in biological product systems (Ayer *et al.*, 2007; Schau and Fet, 2008). One example is in milk based systems, allocation can be based on the energy required to produce milk, maintenance, growing, pregnancy, etc.

Economic allocation and system expansion seem to be the methods that are used most often; however, they both have advantages and disadvantages (Table 9). In studies related to agricultural and food systems, economic allocation is normally the most used co-product handling method (Ayer *et al.*, 2007; Schau and Fet, 2008; Kendall and Chang, 2009), probably because it is a generally applicable method.

Table 9 Advantages and disadvantages of economic allocation and system expansion

	Economic allocation	**System expansion**
Advantages	Generally applicable (Weidema, 1993; Guinee *et al.*, 2004)	Adequate to analyse changes in the product system, demand or production volume (Ekvall and Finnveden, 2001; Cederberg and Stadig, 2003)
Disadvantages	Economic value varies with time (Ayer *et al.*, 2007; Feitz *et al.*, 2007) Tariffs and subsidies make it imperfect (Feitz *et al.*, 2007; Schau and Fet, 2008) Does not represent the effect of decisions (Reap *et al.*, 2008)	Requires more data to include avoided products (Ekvall and Finnveden, 2001; Reap *et al.*, 2008; Thomassen *et al.*, 2008a)

It is commonly affirmed that the choice of method depends on the type of the study: attributional or consequential. Allocation should be used in the first case, while allocation should be avoided by system expansion in the latter (Baumann and Tillman, 2004; Curran, 2007a; Schmidt, 2008b; Thomassen *et al.*, 2008a; Fruergaard *et al.*, 2009). Previously, it had been suggested that system expansion should be used always (Weidema, 2001), and economic allocation had also been perceived as a universal solution (Ekvall and Finnveden, 2001; Guinee *et al.*, 2004).

Different co-product handling methods are one of the reasons for different results and non-comparability among studies of similar products (Azapagic and Clift, 1999b; a; Cederberg and Stadig, 2003; Heijungs and Guinee, 2007; Reap *et al.*, 2008; Cherubini *et al.*, 2009; Flysjö *et al.*, 2011a). However, results with different methods for allocation are not always radically different (Curran, 2007b; Guinee and Heijungs, 2007). In the literature authors sometimes choose to evaluate the sensitivity of results with different allocation or system expansion methods. Essentially the evaluation of sensitivity to co-product handling methodology should be seen as a way to provide robustness to the LCA study process.

Recently, the issue of comparability among studies also has been prompted. Flysjö *et al.* (2011a) analysed different guidelines for co-product handling and has stated the need for harmonization between life cycle approaches studies. This is a critical issue if the results of life cycle studies will be used in communications.

2.2.4.3 Impact assessment

The impact assessment should be planned to achieve the goal and scope of a LCA. This phase involves mandatory and non-mandatory parts. Mandatory parts in ISO 14044 include:

- Choice of impact categories, category indicators and characterization models.
- Classification, which is the assignation of inventory results to impact categories.
- Characterization, which is the calculation of the category indicator results.

The terms used for impact assessment in LCA should be carefully distinguished; Table 10 presents an example of the terms used for the impact category climate change:

Table 10 Example of terms use (ISO, 2006b)

Term	Example
Impact category	Climate Change
Inventory results	Amount of GHG per functional unit
Characterization model	Baseline model of 100 years of the IPCC
Category indicator	Infrared radiative forcing ($W\ m^{-2}$)
Characterization factor	Global Warming Potential (GWP_{100}) for each GHG (kg CO_2 eq./kg GHG gas)
Category indicator result	kg of CO_2 eq per functional unit
Category endpoints	Coral reefs, forest, crops

Typical impact categories are (Baumann and Tillman, 2004):

- Climate change: This refers to the effect of heating the atmosphere due to accumulation of GHG emissions. Most important GHG's are CO_2, CH_4 and N_2O.
- Resources: This refers to the depletion of resources, which may be divided into abiotic and biotic.
- Land Use: this category includes land occupancy and transformation.
- Ozone depletion: The depletion of the ozone in the upper atmosphere as a result of the emission of bromated and chlorinated substances.
- Toxicity: This is used to reflect the toxicological impacts of pollutants; this can be divided into human toxicity and eco-toxicity. The latter can be further divided into terrestrial and aquatic toxicity, and the latter can be divided into fresh and marine toxicity.
- Photo-oxidant formation: Photo-oxidants are created in the lower atmosphere from NO_x and hydrocarbons with sunlight, and can cause human health problems and damage to vegetation.
- Acidification: Acid deposition occurs in some way (acid rain, fog, etc.); resulting in damage to vegetation, buildings and monuments and mortality in aquatic animals. Major pollutants are SO_2, NO_x, HCL and NH_3.

- Eutrophication: This is related to the increased level of nutrients (N and P) which may cause changes in species and biological productivity, degradable organic pollution normally (expressed typically as BOD or COD) also contributes to this category. Major pollutants are NO_x and NH_3.

Non-mandatory parts of impact assessment in ISO 14044 include:

- Normalization, which is the calculation of the category indicator results in relation to reference data in order to understand the relative magnitude of the indicator results. Reference data might be the total indicator result for a country or continent.
- Grouping (of results). This may be used to divide the impacts categories (e.g. local and regional effects, low and high priority, etc.).
- Weighting. This includes the conversion of category indicator results using factors based on societal choices.
- Data quality analysis, which is the analysis of the reliability of the category indicator results.

Grouping and weighting are not based on science, but on value-choices and, thus, results of these parts should be carefully treated.

Databases with ready-made impact assessment methods may be used. Examples of methods are: Eco-indicator'99, EPS, among others.

Eco-indicator'99 is a method intended for designers and engineers in product development applications. It includes normalization factors valid for average damage in Europe and weighting factors based on cultural values in society (Baumann and Tillman, 2004)

EPS (Environmental Priority Strategies in Product Design) is also a method developed for product development in industry. The special issue about this method is that the weighting is based on the willingness to pay to avoid changes in human health, biological diversity, ecosystem production capacity, abiotic resources and cultural and recreational values (Baumann and Tillman, 2004).

2.2.4.4 Interpretation

The final phase is interpretation, in which the significant issues based on all the previous phases are stated, and conclusions and limitations are presented. A check for completeness, sensitivity and consistency is part of the interpretation phase (ISO, 2006b).

2.2.5 Life cycle greenhouse gas emissions of goods and services

Nowadays, life cycle approaches with focus on GHG emissions are common since climate change is a priority regarding environmental policy. Commonly GHG emissions assessment results are called "*Carbon Footprints*". There are several definitions of "*Carbon Footprint*", one is "The total set of GHG emissions caused by an individual or organisation, event or product. It should be expressed in carbon dioxide equivalent (CO_2-eq)" (The Carbon Trust, 2007). Baldo *et al.* (2009) defined the "*Carbon Footprint*" as "the overall amount of carbon dioxide (CO_2) and other greenhouse (GHG) emissions associated with a product along its supply chain, which includes its use phase as well as product end-of-life management". It is mentioned that it should be expressed in CO_2 equivalents, based on the GWP.

The British Standard PAS 2050 for the assessment of life cycle GHG emissions from products (BSI, 2008b) and its guidance document (BSI, 2008a) have recently been developed. The development of these documents has used as reference the ISO standards for LCA and thus they can be perceived as a further specification of generic LCA methodology to focus only on GHG's. The life cycle GHG emissions assessment has in practice 3 phases, the start-up, the calculation of the GHG gas emissions and the next steps.

In the Start-up, the objectives and the functional unit are defined. The usual goal is the reduction of GHG emissions; however, additional objectives can be defined. This phase includes the definition of people and suppliers, resources, schedule and reasons to carry the study.

The calculation of the GHG emissions has 5 steps: construction of the process map, setting of boundaries and priorities, data collection, GHG emissions calculation and an optional step which is the uncertainty check (BSI, 2008b).

Exclusions of the product system are explicitly indicated in PAS 2050. For example, human energy inputs, transport of consumers to and from retail, transport of employees to and from work, animals providing transport and production of capital goods are all excluded from the assessment.

For final disposal a period of 100 years of GHG emissions should be included in the assessment. If carbon storage is supposed to occur in a product, the assessment should include the impact over 100 years, and more than 50 % of the carbon should remain stored for at least 1 year. Therefore, food or feed should not be included as eligible products for storage of biogenic carbon (BSI, 2008b).

There are two types of data: activity data and emission factors. Activity data is the amount of material and energy involved, while emission factors are used to transform activity data into GHG emissions. To calculate the GHG emissions of an activity, the following calculation is used (BSI, 2008b):

$$GHG_{activity} = Activity\ data\ (mass, volume, energy\ or\ distance) \\ \times Emission\ factor\ _{acitivty} (CO_2 eq\ per\ unit)$$

The calculation of the total GHG emissions of a product involves the sum of the emissions for all processes in the product life cycle.

With regard to co-product handling, PAS 2050 (BSI, 2008b) proposes the following hierarchy:

- Allocation should be avoided by process division
- Next option should be system expansion
- If allocation has to be performed, it should be based on economic value of co-products

The first two steps are in concordance with ISO LCA (ISO, 2006b; a), however PAS 2050 sets economic allocation as mandatory once allocation has to be performed.

The last steps are the validation of results, the reduction of emissions and the communication of the results and reductions. Validation is used to verify that the assessment has been in concordance with PAS 2050. It is recommended to have a third party certification when the results are intended to be used for external communication. The reduction of emissions and communication of results are only present in the guidance document and not actually in the standards.

"Carbon Footprinting" has positive and negative aspects in comparison to a "full" LCA. In particular positive aspects are related to its huge popularity (especially in UK) and potential for increasing consumer awareness of the environmental impact of products (Weidema *et al.*, 2008a). It should be noted that the term "Carbon Footprint" is just a new name for "the result of the life cycle impact category indicator global warming potential", which has been around for decades as part of impact assessment in LCA (Finkbeiner, 2009).

The development of specific standards for GHG emissions may be considered unnecessary because the full LCA standards already cover climate change issues (SETAC Europe LCA Steering Committee., 2008; Weidema *et al.*, 2008a). Furthermore, Schmidt (2008a) considered that focusing only on the "Carbon Footprint" is a backward step, since there are three key aspects to sustainability: environmental, economic, and social. With development of the "Carbon Footprint" concept even the environmental is replaced by one single indicator (diminishing the importance of other environmental impacts).

Of the different types of sustainability assessment tools the product-related tools are the best suited to assess in a direct manner the environmental impact of human created systems.

2.2.6 Comparative life cycle assessment

A comparative LCA is a “LCA study in which two or more alternative product/services are compared”. In comparative studies, the functional unit definition and methodological choices for different product/services should be comparable (Baumann and Tillman, 2004).

System boundaries and co-product handling procedures should be carefully defined for the product systems that are being compared. This seems to be a limitation for the comparison of LCA results among studies, however it is possible. For some products, it has been shown that general trends can be used to compare results from different studies (de Vries and de Boer, 2010). Comparative LCA can be used in a business to business approach to include environmental criteria in the selection of providers or in a business to consumer approach to give the opportunity to the public to select products with information about their environmental profile.

2.3 Climate change and the rendering industry

2.3.1 Legislation

The Climate Change Act (UK, 2008a) sets legally binding targets for the UK to reduce its GHG emissions by 80% by 2050 and 34% by 2020 in relation to the baseline year (1990 – 1995). These targets should be met by action within the UK and abroad.

The Energy Act (UK, 2008b) has as key aspects the Renewable Obligation (RO) and the Renewable Heat Initiative. The first one is related to the higher inclusion of large scale renewable electricity in UK and the latter is related to establish financial support towards the use of heat from renewable sources. Heat technologies included are: heat pumps, biomass boilers, solar-thermal water heaters and combined heat and power (CHP) plants which use renewable fuels (DECC, Not dated-b).

Also part of the Energy Act, and alongside with the RO, the Feed-in Tariff system is designed to encourage the development of small scale (less than 5 MW) low-carbon electricity generation, particularly by organisations and individuals that have are not in the electricity market (DECC, Not dated-a).

Among the instruments already in place regarding GHG emissions from industry in UK legislation is the Climate Change Levy, which is a tax on energy use from fossil fuels (DEEC, Not dated). For example, the use of natural gas is taxable for £0.00154 per kWh. The rendering industry is required to pay this levy when using fossil fuels.

Given the UK ambitious mitigation targets, further and stricter measures and policies are likely to be introduced in different sectors in the short to medium term. It is likely that the rendering industry (as any other sector) will be challenged to reduce its GHG emissions. Therefore, it is important to identify which processes in the production chain are the most important regarding GHG emissions.

2.3.2 Competitiveness

Green marketing has been discussed since the early 1990s. The aim of the field is to include environmental aspects in marketing. It is supposed that if the environmentally friendly characteristics of products are shown to the consumer by means of for example "ecolabels", consumers would choose products with "greener" characteristics (Rex and Baumann, 2007).

There are three types of ecolabels; I, II and II. Ecolabels type I provide information about the environmental friendliness of a product in a specific product category. This type requires third party authorization. Ecolabels type II are self-declared environmental claims that do not require certification (Baumann and Tillman, 2004).

Ecolabels type III, also called Environmental Product Declarations (EPD), are based on LCA results to be communicated in market situations. LCA methodology for EPDs should be very strictly standardized since they are supposed to be used in comparison of products (Baumann and Tillman, 2004). It seems like EPDs of full LCA (ecolabels type III) are best suited for the communication between businesses, while ecolabels of inferior types are well suited for communication to consumers. It should be noted that traditional

LCA results are not necessarily sufficient to ensure complete environmentally sound practice, however they can be a central part of an ecolabelling system (Mungkung *et al.*, 2006).

In the UK, the Carbon Trust (The Carbon Trust, 2010b) has established the Carbon Trust Footprinting Company to provide certification of GHG LCA performed according to BSI PAS 2050. The Carbon Footprinting Label has been developed to be used in both business-to-consumer and business-to-business situations (The Carbon Trust, 2010a). This is probably an important characteristic of "Carbon Footprinting", its simplicity. In contrast, the result of a full LCA might not be usable for communications with final consumers as they might be too complex.

Rendered products are used in some applications where marginal sources of protein and fat sources are used. For example, an alternative source of protein with a similar content of crude protein would be soya bean meal which has 516 g per kg dry matter, while PAP has 538 g per kg dry matter (Sellier, 2003). Fat alternatives can be selected based on chain length, tallow and palm oil are long chain fatty acids (C_{16-18}) (Postlethwaite, 1995). However, there are a number of other proteins and oils that might also be of interest. Additionally there might be competition between rendering products themselves, different proportion of fuels used in different plants or different sources of ABP (cattle, lamb, pigs, poultry, fish, etc). In the context of the rendering industry, its products are always used in other business sectors.

Comparison of the LCA results for different products or the same product from different providers that can be used in the same application might help the selection of providers. This is especially important in a future world with high demand and legislative pressure for even more climate change friendly products.

2.3.3 Greenhouse gas emissions from rendered product systems

In rendered product systems (before LCA boundaries are set), there are multiple sources of GHG's (Figure 21). The rendering industry consumes significant amounts of energy to release fat, evaporate water and sterilise the raw materials (Kalbasi-Ashtari *et al.*, 2008).

In rendering plants, the main source for GHGs is the combustion of fossil fuels for process heat.

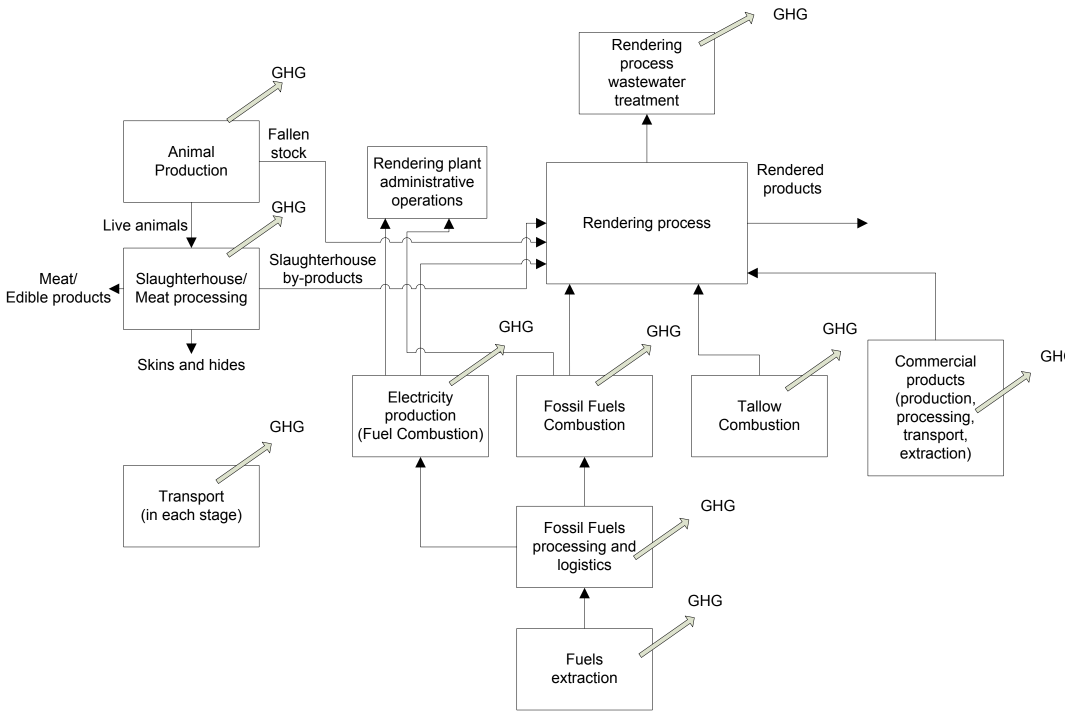

Figure 21 Identification of GHG emission sources from the rendering products system (without setting boundaries)

Category 1 rendered fat is also burnt in different proportions to produce process heat. However, rendered fat is biogenic and thus the CO_2 produced in its combustion can be considered part of the natural carbon cycle. Bioenergy carbon neutrality is based on the assumption that all the CO_2 that biomass absorbs from the atmosphere while growing will be released as CO_2 by decomposition (naturally) or by combustion (artificially) and thus forms part of the natural carbon cycle. The carbon cycle regulates the content of CO_2 and CH_4 in the atmosphere (Danny Harvey, 2000). Carbon in plant biomass is carbon removed from the atmosphere by photosynthesis. Plant, soil and animal respiration and decomposition complete part of the carbon cycle in the form of CO_2 (or as CH_4 if converted under anaerobic conditions) (Denman *et al.*, 2007). In contrast emissions of CO_2 resulting from the burning of fossil fuels result in a net gain of CO_2in the atmosphere, consequently contributing to global warming.

It is important to notice that the net balance occurs when it is CO_2 which is absorbed (during photosynthesis) and released under aerobic conditions (respiration or combustion) in a relatively short term cycle. CH_4 from decomposition of biomass in anaerobic conditions is biogenic; however it does represent a net gain of CO_2 equivalents in the atmosphere as the GWP of CH_4 is 25 times that of CO_2 (Solomon *et al.*, 2007a). When the biomass that is burnt is part of the long term reservoirs of carbon in Biota, the combustion of that biomass produces biogenic CO_2 that cannot be considered neutral regarding its effect to the accumulation of CO_2 in the atmosphere as it is associated with a natural reservoir of carbon. This is critical for emissions associated with land transformation. In summary, not all biogenic emission of carbon can be considered Climate Change neutral.

Emissions from processes actually outside the rendering plant are also presented in Figure 21. Livestock production (and animal slaughtering) results in GHG emissions (de Vries and de Boer, 2010).

The production of electricity that is supplied via the national grid involves GHG emissions associated with the burning of fossil fuel; however this is not the end of the story in a life cycle perspective. Fuel extraction and processing and logistics also have associated GHG emissions. Fossil fuels are used for electricity production outside of the rendering plant (i.e. by power stations) and for heat production inside the rendering plant.

The production, processing, transport, extraction of raw material and associated energy production of commercial products (e.g. chemicals) used in the rendering process also produces GHG emissions.

Finally, there is transport between almost every stage: animal production to slaughterhouses/meat plants, and meat-plants to rendering plants. Additionally there is transport associated with feed, fuels and commercial products. Transport is mostly based on fossil fuel powered vehicles, and thus there are GHG emissions associated with it.

2.3.4 System boundaries in LCA of rendered products

Figure 21 illustrates the entire rendered product system before system boundaries have been established. ABP are produced in the animal production system and can be perceived as a waste or a valuable product. In guideline documents for LCA the definition of a waste is:

- "substances or objects which the holder intends or is required to dispose of" (ISO, 2006b)
- "materials, co-products, products or emissions which the holder discards or intends, or is required to, discard" (BSI, 2008b)

These two definitions are in agreement, however they are qualitative and in the context of animal by-products are not necessarily operational (ABP are by-products that the animal and meat producers are "required" to dispose of). Therefore in the current work, the recommendation by Guinee *et al.* (2004) to differentiate between co-products and wastes has been used, and consequently a waste is considered an economic flow with null or negative cost. In the context of animal by-products if the holder (farm or slaughterhouse) paid for its disposal, than they were considered wastes, which do not incur the environmental impact associated with their production. Whilst when the holder (farm or slaughterhouse) received revenue in exchange of the animal by-product material, than it was considered a co-product, that based on an adequate co-product handling approach incurs an environmental burden associated with its production.

Based on the UK ABP industry there are 3 main groups of ABP. Category 1 (including category 2 material) mammalian material, category 3 mammalian material and category 3 poultry material. Category 3 mammalian material includes ruminants and pork in the UK. This is not necessarily valid for other countries; it seems that in some other countries there are rendering plants processing pig material exclusively (pers comm Stephen L Woodgate 2011).

The price of slaughterhouse by-products for mammalian material in the UK is presented in Table 11. The reference provides a division for prices based on two categories of weekly amounts, 10 – 50 tonnes and over 50 tonnes. Greater amounts have "higher" price. Only the slaughterhouse output described as "Best fat" has a positive price. Most of the mammalian derived slaughterhouse by-products in the UK in the periods presented represented costs to the meat producers. Transport has not been included in the figures.

Table 11 Average price of red meat derived slaughterhouse by-products (£/tonne) (Meat Trades Journal, 2007-2009)

MTJ classification	weekly amount category	10 - 50 tonnes Dec 2006 - Sep 2009	10 - 50 tonnes 2008	over 50 tonnes Dec 2006 - Sep 2009	over 50 tonnes 2008
Best fat	3	13.4	14.4	77.8	82.7
Other fat	3	-2.7	-1.1	14.1	16.6
Bones	3	-75.0	-80.9	-55.4	-55.8
Hard Offal	3	0.0	0.0	0.0	0.0
Other Offal	3	-79.5	-84.0	-54.7	-61.9
SRM	1	-78.0	-81.3	-56.1	-58.3

Using the prices in Table 11, and detailed partition of mass for cattle parts in EBLEX (2006) it is possible to construct the economic value distribution of the main economic flows from the slaughtering of beef cattle (Table 12). Both categories of mammalian ABP (1 and 3) are negative economic flows and therefore have been treated as wastes throughout the research programme presented.

Table 12 Mass of beef slaughtering streams in kg for ideal and worst case based on an average beef animal adapted from EBLEX (2006) and economic value in percentage based adapted from Meat Trades Journal (2007-2009)

Slaughtering output economic flows	Best case Mass (kg)	Best case Economic value (%)	Worst case Mass (kg)	Worst case Economic value (%)	Positive economic flow
carcass lean	192.54	96.3%	192.54	96.8%	Yes
edible material	122.61		65.76		Yes
hide and skin	42.49	4.2%	42.49	5.2%	Yes
petfood	4.91	0.1%	0		Yes*
gut content	89.45	0.3%	0		Yes*
category 3 ABP	64.46	-0.4%	123.98	-0.4%	No
category 1 ABP	65.403	-0.5%	175.21	-1.5%	No
Total beef cattle mass	581.863		599.98		

*They would have a positive price in the ideal case defined by the reference; they do not necessarily exist currently.

In contrast Table 13 presents the average prices of poultry derived ABP. Poultry ABP (over 50 tonnes) are economic flows of positive value and therefore will be treated as valuable by-products throughout the research programme presented.

Table 13 Average price of poultry derived slaughterhouse by-products (Meat Trades Journal, 2007-2009)

MTJ classification	weekly amount	10 - 50 tonnes		over 50 tonnes	
	category	Dec 2006 - Sep 2009	2008	Dec 2006 - Sep 2009	2008
Carcasse	3	0	0	23.0	12.1
Offal	3	0	0	13.3	4.8

2.3.5 Type of life cycle assessment and main methodological choices

The main purpose of this research programme is to present attributional LCA results of GHG emissions of rendered products. Therefore methodological approaches associated with attributional LCA are used.

The main co-product handling approach used is allocation (either based on economic or mass ratio of co-products). However, system expansion is used also when the Climate Change effect of the ABP processing system is evaluated in the context of meat production and for waste disposal. In these cases the consequential loops associated with the by-product or waste streams are included in the system according to the availability of data of the involved processes.

Data is used for average production and technology, however in some cases sensitivity is tested with marginal technologies (in particular in the case of co-production of electricity).

In the case of the production of animal feeds, emissions associated with land transformation associated with soya bean are included as presented in Ecoinvent databases (Ecoinvent Centre, 2010). Land use change associated with soya bean is modelled including the area of land transformed from tropical, shrub and arable land associated with soya bean cultivation in the previous 5 years. The 5 year average includes cultivation in the average transformed land area.

2.4 Conclusion

The rendering industry processes a significant amount of biomass. With these materials, renderers produce rendered protein meals and fats, which depending on the category material can be useful or have to be disposed of. Category 3 rendered products are useful as pet food ingredients, organic fertilizers, and as raw material in the oleochemical industry. Category 1 rendered products are required to be disposed by combustion (with energy recovery) or can be used as feedstock for biofuels.

The environmental burden of a process should be allocated between the various co-products, but not the wastes. When ABPs do not have a positive economic value, the environmental burden of their production should be ascribed to the animal products. Arguably as they are unavoidable by-products, their processing should be included as part of animal food production. Energy recovery from ABPs within or outside the rendering industry has relevance for GHG emissions since they are biomass and biogenic CO_2 emissions can be considered carbon neutral. However in the production of ABP, the most important GHG emissions are CH_4 and N_2O (from animal production) which are not climate neutral. Therefore the co-product handling methodology and system boundaries are critical in the calculation of GHG emissions from rendered products.

The organisation of the current thesis is presented in Figure 22. Chapter 3 presents an overview of publicly available results of GHG emissions of animal and meat production which is where animal by-products are produced. The primary objective of this study was to calculate the GHG emissions of rendered products in the UK (Chapters 4 and 7). As shown in the previous section, beef and poultry ABPs can be treated differently as the former represents a cost and the latter represents a positive value, therefore the GHG emissions of the production of broilers is investigated in Chapter 6, and the processing of poultry meat and poultry by-products is investigated in Chapter 7. The effect of the inclusion of ABP processing as part of the meat production system was also investigated (Chapters 5 and 7). Ultimately, the evaluation of rendered products as feed ingredients for farmed animals is performed with salmonid feed production as a case study (Chapter 8).

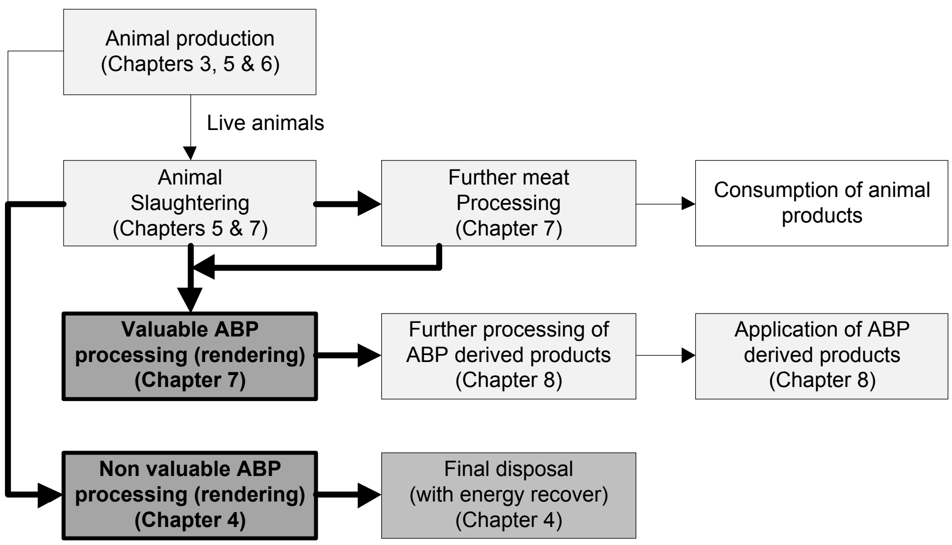

Figure 22 The animal by-product system (chapter focus in parentheses)

Chapter 3

Greenhouse gas emissions of land based meat production systems: An overview

3 Greenhouse gas emissions of land based meat production systems: An overview

3.1 Abstract

The environmental burden of food production has been an objective of major scrutiny in recent decades. Greenhouse gas (GHG) emissions from animal production are an important contributor to Global Warming. Animal production (as any other agricultural activity) is predominantly associated with CH_4 and N_2O GHG emissions that are associated with enteric fermentation, manure management and fertiliser application. Recently (especially in the last decade), studies on the GHG emission intensity of animal products have been increasingly completed. Studies have taken regional or country level intensity or process based life cycle assessment approaches.

The objective of this study was to provide an overview of 34 studies that have investigated the GHG emissions of land-based meat production systems to understand major methodology issues and to compare and contrast the results presented in these studies. Studies were grouped by major land based meat production systems (beef, lamb, pigs and poultry). There is agreement amongst studies that ruminant derived meat is associated with the highest GHG emission intensity. Poultry systems appear to have lower GHG emission intensity than pig systems.

In general, calculation of GHG emissions intensity from land based meat production is in a mature state. However, there is need for further work in the treatment of emissions associated with the consequences of changes. In particular for the treatment of GHG emissions associated with land transformation (forests to arable land for feed production or to pasture land).

3.2 Keywords

Livestock, sustainability, life cycle assessment, intensity, carbon footprint, review

3.3 Introduction

Land based animal production (dairy, meat and eggs production) is an important economic sector accounting for 40% of the agricultural gross domestic product, providing one-third of human's protein intake and employment for 1.3 billion people (Steinfeld *et al.*, 2006). However it also contributes 18% to anthropogenic global greenhouse gas (GHG) emissions; this is higher than the share of transport and represents 80% of agricultural emissions (Steinfeld *et al.*, 2006). Globally, the major GHG is CO_2, mostly associated with the combustion of fossil fuels to produce electricity, heat and transport (Barker *et al.*, 2007). Emissions from agriculture are mostly associated with other more powerful GHG, namely CH_4 and N_2O. The former is mainly associated with anaerobic decomposition (in enteric fermentation and manure management) and the latter mainly associated with fertiliser application and manure management.

The climate impact of livestock/meat production is exacerbated by increasing production trends over the past decade. Globally land based meat production increased from approximately 233.5 to 283.9 million tonnes (FAO, 2011) between 2000 and 2009, corresponding to a 21.6% increase. Total meat production in Europe increased from approximately 51.7 to 54.8 million tonnes in the same period (representing a 6% increase). Annual meat production is projected to increase to 465 million tonnes by 2050, with the major increase in developing countries (Steinfeld *et al.*, 2006). Consequently there have been a substantial number of studies published that have investigated the GHG emissions of animal production. The approaches used to calculate GHG emissions have varied from using production statistics to calculate regional intensities to application of standard life cycle assessment (LCA) methodology. LCA is a mature tool to evaluate the environmental burden of a product throughout its life cycle standardised by ISO (ISO, 2006b; a). LCA has been applied extensively to food production (Roy *et al.*, 2009; de Vries and de Boer, 2010; Cerutti *et al.*, 2011; Henriksson *et al.*, 2011; Milani *et al.*, 2011).

LCA studies on meat production have been performed to evaluate the different environmental impacts associated with their production (e.g. Climate Change, Acidification, Eutrophication, Land use, Ecotoxicity, etc.). In recent years part of LCA research has focused only on the evaluation of the Climate Change impact of a product or service; the results of this are commonly referred to as *Carbon Footprints* (BSI, 2008b; a; Weidema *et al.*, 2008a). *Carbon Footprint* has positive and negative attributes in comparison to a "full" LCA. In particular positive aspects are related to its huge popularity (especially in UK) and potential for increasing consumer awareness of the environmental impact of products (Weidema *et al.*, 2008a). In contrast, focusing only on the *Carbon Footprint* can be considered as a backward step, since there are three key aspects to sustainability (environmental, economic, and social) and with the development of the *Carbon Footprint* concept even the environmental is replaced by one single indicator (Schmidt, 2008a). It seems that the future of LCA is actually in the direction of integral sustainability assessment with the associated broadening of the impact assessment to include not only the environment but also social and economic dimensions (Guinee *et al.*, 2010). However, Climate Change can be considered the greatest sustainability threat that humanity is currently facing. The potential severity of the consequences should encourage the mitigation of GHG emissions by any means (New *et al.*, 2011). Therefore focusing on only Climate Change may be useful in the current world. GHG emission mitigation opportunities in animal production have been presented in Weidema *et al.* (2008b) and de Boer *et al.* (2011).

Beef, lamb, pig and poultry meat represented 22-23%, 5%, 37% and 32% of global land based meat production (summing up to 98%) respectively between 2005 and 2010 (FAO, 2011). The purpose of this study was to contrast methodological choices and GHG emissions reported by 34 studies completed on the production of land based meat.

3.4 Scope of the review

The 34 studies presented in this review have been grouped as beef (Table A-1), lamb (Table A-2), pig (Table A-3) and poultry (Table A-4) in Appendix A. The focus of the current review is on meat production systems, and consequently other animal products such as dairy and eggs are not included.

The term poultry is used instead of chickens because one of the studies also reports results for turkeys. Some studies do not explicitly state the use of LCA or *Carbon Footprint* methodology; however they studied the GHG emissions intensity of meat production systems and therefore were included in the review. This review is mainly focused on peer-reviewed literature; however some institutional reports (frequently cited in peer reviewed literature) are also included.

Some of the studies cover more impact categories than Climate Change; however the focus of this review was limited to this impact category. To enable direct comparison between studies, only studies with results that could be presented in units of kg of either carcass weight (CW) or live weight (LW) were included, for example some studies (Flessa *et al.*, 2002; Stewart *et al.*, 2009) were not included as they reported results for integrated farming without presenting results for beef LW or CW.

Studies were classified according to: geographical coverage, functional unit (FU), approach to economic and natural flow calculation, co-product handling, scope and GHG emission sources included, and results in GHG emission per FU (Appendix A). Some single references report results for more than one meat system and therefore have been included in more than one table (Williams *et al.*, 2006; Weidema *et al.*, 2008b; Cederberg *et al.*, 2009a; Edward-Jones *et al.*, 2009; Peters *et al.*, 2010; Lesschen *et al.*, 2011; Phong *et al.*, 2011).

3.5 Summary of differences among studies

3.5.1 Geographical coverage

The current review includes 20 studies on beef production (Table A-1 in Appendix A). Ten of these studies were for Europe or European countries, seven for North American countries (one including a comparison with production in Sahelian Africa) and one each for Japan, Australia and Brazil.

As a first observation, lamb production studies are not as numerous as for all the other meat production systems (4 studies in Table A-2 in Appendix A). This may be because lamb meat production is comparatively low, accounting for only 5% of global meat production in the period 2005-2010. All the studies found for lamb meat production were for the UK and Australia which perhaps reflects the importance of this meat in these countries in comparison to other regions. Production in the UK accounted for 32% of the total production of lamb meat in the EU between 2005 and 2010. UK and Turkey were the top producers in the EU in the period. UK was the 11th greatest producer in the world in the period. Australia and New Zealand were the third and fourth lamb producers in the world after China and India (FAO, 2011).

Table A-3 (Appendix A) presents the 12 studies for pig meat included in this review. Table A-4 presents the studies on poultry meat production (9 studies). Poultry and pig meat production are the greatest land based meat production systems (FAO, 2011), and therefore it seems that the interest for understanding their environmental relevance has increased lately. Studies are mostly being performed for systems in Europe and North America.

LCA results for meat production systems vary depending on geographical coverage. This can be attributed to variations in animal husbandry practices and the natural flows associated with inputs and activities (for example, the GHG emissions intensity of energy systems in different countries can vary considerably). Studies from developing countries seem scarce; however this is not an issue necessarily exclusive of studies on GHG emissions from meat production systems, but maybe with research into environmental issues in general.

3.5.2 Scope and GHG emissions sources

The studies included in the current review were grouped into three “scopes” according to the processes included: These scopes were cradle-to-farm-gate, cradle-to-slaughterhouse-gate (2 studies (Dalgaard *et al.*, 2007; Peters *et al.*, 2010)), and cradle-to-grave (1 study (Weidema *et al.*, 2008b)). Most studies (31 studies) applied a cradle-to-farm-gate approach, including only processes associated with animal production. Cradle-to-slaughterhouse-gate includes cradle-to-farm-gate processes with the addition of transport to slaughterhouse and the inputs and outputs associated with slaughtering. Cradle-to-grave includes cradle-to-slaughterhouse-gate with the addition of consumption and final waste management.

All the studies on beef and lamb (Tables A-1 and A-2 in Appendix A) have included GHG emissions from: feed production, enteric fermentation, on-farm energy use, manure management and soil management. In most studies feed production includes GHG emissions associated with feed (crop) production, processing, and delivery. In some studies feeds are produced or assumed to be produced on the farm and consequently there are no GHG's associated with transport (Phetteplace *et al.*, 2001; Beauchemin *et al.*, 2010). This is also obvious for systems that do not include feedlot operations (Subak, 1999; Cederberg *et al.*, 2011). Lesschen *et al.* (2011) included feed production; however they did not include feed processing or delivery.

Not all the studies explicitly state whether indirect emissions associated with soil and manure management are included, however it seems that these processes are normally included. There are studies that explicitly stated that indirect emissions were included (Phetteplace *et al.*, 2001; Williams *et al.*, 2006; Cederberg *et al.*, 2009a; Edward-Jones *et al.*, 2009; Beauchemin *et al.*, 2010; Nguyen *et al.*, 2010a; Pelletier *et al.*, 2010b; Veysset *et al.*, 2010; Beauchemin *et al.*, 2011; Cederberg *et al.*, 2011; Eady *et al.*, 2011; Foley *et al.*, 2011; Lesschen *et al.*, 2011).

Pig systems are different to ruminant systems. Ruminants rely more on pastures, while pigs normally are fed with feeds produced from outside the farm. Apart from this and its associated soil management, the rest of the processes are the same to those in ruminant meat systems. All the reviewed studies on pig meat systems included: feed production, enteric fermentation, energy use, manure management and soil management.

Poultry meat production systems are very different to ruminant systems. Similarly to pigs, birds are monogastric animals, however from a GHG emissions perspective they are different. Cradle-to-farm-gate poultry meat studies normally include feed production (and delivery), on-farm energy use and manure management. Enteric fermentation is minimal in birds and therefore is has never been included.

Weidema *et al.* (2008b) is the only study with a cradle-to-grave approach that includes all the processes associated with meat production, and therefore animal by-products processing appears to be included. Peters *et al.* (2010), one of the two cradle-to-slaughterhouse-gate studies, explicitly stated that slaughterhouse processes are included; however it is not clear how animal by-products processing was treated. Dalgaard *et al.* (2007), the other cradle-to-slaughterhouse-gate study explicitly stated that the processing of animal by-products was included.

3.5.3 Functional unit

The definition of the functional unit is one of the first steps in LCA, and is the basis for calculation of economic and natural flows in the product system (ISO, 2006b; a). The main choices of functional unit in studies that have a cradle-to-farm-gate approach are either CW or LW. LW is the actual weight of the live animals leaving the farm gate. When based on CW, the carcass yield (sometimes called the killing-out-percentage (KoP)) is used to quantify results based on the "saleable part" derived from the slaughtering process. Yields used may vary from study to study (Tables A-1, A-2, A-3 and A-4 in Appendix A). Some studies have used a top-down approach to calculation using statistics based on CW. In principle, it seems that cradle-to-farm-gate studies (i.e. not including post-farm stages) processes are better based on LW, as this is the actual outcome of the farm gate. Although to provide results based on CW seems useful since the main purpose of the meat production system is to produce actual edible meat, it does not seem ideal to present the results based on the outcome of the slaughtering process when the complete inputs and outputs of the slaughtering process are not included. The difference between LW and CW is approximately the mass of animal by-products and their processing is not taken into account in cradle-to-farm-gate studies.

3.5.4 Approach to economic and natural flows calculation

The approach to calculation of economic and natural flows varies from study to study. Some studies used top-down approaches (national inventories and/or input-output tables) as main data sources, whilst some studies used a bottom-up approach based on real process data from farms. In addition, simulation has been used to model whole farms or entire country level systems. Studies that combine bottom-up with top-down approaches to calculation can be called “hybrid” studies in contemporary LCA terminology (Weidema *et al.*, 2008b; Cederberg *et al.*, 2009a). It is important to realise that some studies may be in fact “hybrid” studies although it is not necessarily explicitly stated.

Emissions from enteric fermentation, and soil and manure management has been quantified by IPCC emission factors or methodologies (IPCC, 2006) in most studies. The use of life cycle inventory databases is especially relevant for the inputs to the farm (e.g. feeds and energy). Some studies however have also included the modelling of the production of feeds (at least partially); this is obvious for studies where the feeds (or part of them) are produced on the animal production farm.

3.5.5 Co-product handling

Co-product handling is probably the most debated issue in LCA methodology (Weidema, 1993; Azapagic and Clift, 1999b; Ekvall and Finnveden, 2001; Ayer *et al.*, 2007; Curran, 2007a; Reap *et al.*, 2008; Flysjö *et al.*, 2011a). In principle, at least for cradle-to-farm-gate studies of pure meat systems there seem to be only two multiple output problems: the farm outputs (animals, manure) and the production of feeds (or any other input) from multiple output systems (e.g. soya bean and soya bean oil from the processing of soya beans).

Most cradle-to-farm-gate studies reviewed here used main or primary product (also called no-allocation) for the output of the farms. Some studies based on LW stated that co-product handling (e.g. allocation) is not required as the animals leave the farm as single units; however they included feed production sometimes from databases that normally required previous allocation (therefore their system may actually include the use of a co-product handling approach although it is not necessarily stated). Manure management is normally handled with system expansion, although this can also be seen as the usual and reasonable way of including waste management in LCA.

The inclusion of slaughterhouse processes (cradle-to-slaughterhouse-gate studies) adds an additional complication as meat processing systems produce a great variety of co-products and by-products (e.g. edible carcass, skins and hides, different types of slaughterhouse by-products). The only cradle-to-grave study (Weidema *et al.*, 2008b) used input-output tables to account for all the environmental impacts of meat production in the economy and therefore animal by-products should be included.

Some LCA studies of beef (included in Table A-1 in Appendix A) are also associated with milk production. These and other studies have found important implications of the link between milk and beef systems (Cederberg and Stadig, 2003; Williams *et al.*, 2006; Weidema *et al.*, 2008b; Cederberg *et al.*, 2009a; Nguyen *et al.*, 2010a; Flysjö *et al.*, 2011a; Flysjo *et al.*, 2012). It has been shown that beef from systems that include dairy calves have lower resource consumption and environmental impact than 100% beef systems (Williams *et al.*, 2006; Weidema *et al.*, 2008b; Nguyen *et al.*, 2010a). In fact some studies have shown that the environmental benefits of increasing milk yield with the consequential reduction in the number of dairy cattle is offset since more 100% beef systems are required to cover the beef demand (Weidema *et al.*, 2008b; Cederberg *et al.*, 2009a; Nguyen *et al.*, 2010a). The reason is quite obvious; since the environmental burden associated with pregnancy of cows can be divided for both milk and beef in milk-beef systems. The co-product handling of milk and animals for different purposes (dairy or beef) is a very critical issue in milk-beef systems, which are common in Europe.

Studies based on lamb present the same characteristics to those of beef (at least of pure beef systems). Lamb studies do not have the co-production issue of milk, however they do have a co-production issue regarding animals and wool. Lamb systems appear not to be based strongly on feedlots and therefore the issue of co-product handling in feed production seems not very important. Manure management presents the same implications as beef systems. Three studies on lamb meat used economic allocation to divide the life cycle inventory between lambs and wool.

Co-product handling issues that can be seen in pig systems are: the co-production of porker and sow (Basset-Mens and van der Werf, 2005), the co-production of feed ingredients (with special relevance for soy bean meal) and the handling of manure. A special case is the study by Phong *et al.* (2011) which studied integrated aquaculture-agriculture producing some different products (rice, fruits, vegetables, pigs, poultry, and fish), and therefore requires co-product handling for different animal and vegetable species. Of the reviewed studies on pig meat, two (Dalgaard *et al.*, 2007; Nguyen *et al.*, 2010b) used a consequential approach that included the consequences of co-production in some feed ingredients (e.g. soya bean meal) through system expansion. The remaining studies used an attributional approach (based in allocation approaches).

Co-product handling issues in cradle-to-farm-gate poultry systems seem to be associated with feed production and manure management. In addition when using top-down approaches to calculation as in Verge *et al.* (2009b) and Lesschen *et al.* (2011), it seems that a co-production issue arises as the statistics for broilers and culled laying hens are commonly amalgamated as poultry meat. However this does not seem to be a critical issue as the amount of meat from culled laying hens do not seem important in comparison with broiler meat. For example in Canada at least, the total LW of culled laying hens was 5% of the total LW (Verge *et al.*, 2009b).

3.5.6 Greenhouse gas emissions

Results for GHG intensity of beef production in Europe vary between 7 and 49 kgCO_2 / kg LW (Table A-1 in Appendix A), using mid, averages or conventional figures when ranges or various results are reported, when results are reported based only in CW (and the carcass yield is not reported) it has been assumed as 58%. The maximum is for European generic production from 100% beef systems when including the opportunity cost of land (Nguyen *et al.*, 2010a). North American studies, normally based on regional level or generic practices, present more similar results raging between 8 and 19.2 kgCO_2 / kg LW.

Enteric fermentation, soil and manure management are normally the most important contributors in the GHG emissions from beef production systems. Differences in the contribution from these sources can be attributed to different economic flows and also to the range of emission factors reported in the literature. The GWP of CH_4 and N_2O used to characterise these GHG may vary between studies, however this would not result in a significant variance in the results obtained. A case study of the differences in using different range of emissions factors has been described by Edward-Jones *et al.* (2009). For example, in the study for Sweden completed by Cederberg *et al.* (2009a) and involving a hybrid approach, enteric fermentation accounted for 55%, manure management and application for 19%, feed production and delivery accounted for 23%, and indirect N_2O for 2%of the total GHG emissions of beef. In contrast, enteric fermentation accounted for 17% and 44% for two different farms in Wales using a bottom-up approach (Edward-Jones *et al.*, 2009).

GHG emissions from land transformation is a relatively new issue in LCA of meat products (and in LCA in general), although it was included in an early study by Subak (1999). Two studies deal with land transformation in detail (Nguyen *et al.*, 2010a; Cederberg *et al.*, 2011). Nguyen *et al.* (2010a) studied two issues: the opportunity cost of land associated with the loss of carbon sequestration potential in beef systems in Europe, and the increased demand of land for the production of feeds (in particular soy bean produced in South America). They found that depending on the role of grasslands and croplands, land used related emissions could be positive or negative. Highly and moderately productive grasslands acted as carbon sink, whilst extensive use of croplands acted as a carbon source. In their 100% beef cattle case, GHG emissions were 27.3 kg CO_2e / kg CW and 84.1 kg CO_2e / kg CW when land transformation was not and was included, respectively (a threefold increase). Most studies of European beef production have assumed a carbon balance in soil. Cederberg *et al.* (2011) considered the land transformation of tropical rainforest to pasture land in Brazil, and reported the alarming figure of 726 kg CO_2e / kg CW for beef produced in newly deforested land using 20 years for the production period over which the emissions from the initial deforestation are amortized (the shorter the period the higher the emissions). In Cederberg *et al.* (2011), enteric fermentation and manure management were minor relative contributors (although in magnitude similar with other studies) in comparison to the GHG emissions from land transformation. These studies basically included marginal production issues. Nguyen *et al.* (2010a) studied marginal production of feed ingredients (in South America) and the consequences of potential land transformation in Europe, whilst Cederberg *et al.* (2011) studied the consequences of the marginal production of beef, taking into consideration that Brazil is the top exporter of beef and that the associated growth is being driven by exports. The contrast between marginal and average production is associated with the contrast between consequential and attributional LCA, which are important issues in LCA studies (Dalgaard *et al.*, 2008; Thomassen *et al.*, 2008a; Finnveden *et al.*, 2009), and it seems that they are extremely critical issues in LCA of livestock production.

Results for GHG intensity of lamb production vary considerably from 7.5 to 51.7 kgCO_2 / kg LW for cradle-to-farm-gate studies (Table A-2 in Appendix A). It should be noted that the high extreme is from the study by Edward-Jones *et al.* (2009), which is based on one case study with real data from a farm producing beef and lamb. Similar to beef systems, enteric fermentation, and soil and manure management are normally the most important contributors in the GHG emissions from lamb production systems. Differences in the contribution from these can be attributed to different economic flows and also to the range of emissions factors in the literature. The differences in using different emissions factors has been studied by Edward-Jones *et al.* (2009).

Results for pig meat systems range between 1.6 and 15.7 kg CO_2/ kg LW for cradle-to-farm-gate studies. The highest result is from the only study including emissions associated with land transformation in feed production and the opportunity cost of land (Nguyen *et al.*, 2010b). When not including the extremes, GHG emissions per live weight are lower than for ruminant systems with emissions ranging from 1.5 to 4.9 kg CO_2e /kg LW for Europe. GHG emissions from pig systems in North America are in a similar range with 2.3 to 4.5 kg CO_2e / kg LW. The study by Phong *et al.* (2011) is a very different case for multiple output farms in the Mekong Delta.

The GHG emissions contribution from the different processes can vary from study to study. The contribution of enteric fermentation to the GHG emissions of pig production is considerably lower than in ruminants. In addition, the effect of previous generation is lower than in ruminants. The contribution from different sources in the different pig systems studied is similar. In the base case in Nguyen *et al.* (2010b), feed production and delivery accounted for 61% of the GHG emissions, on-farm CH_4 and N_2O emissions (manure management and enteric fermentation) for 35%, and on-farm energy use for 5% (only counting positive GHG emissions, manure land spreading provides credits from the avoidance of inorganic fertilisers production). In general, emissions associated with feed production and manure management seem to be the most important contributors (Basset-Mens and van der Werf, 2005; Dalgaard *et al.*, 2007; Cederberg *et al.*, 2009a; Pelletier *et al.*, 2010a; Stone *et al.*, 2012).

GHG emissions from poultry meat systems seem to range between 1 and 2 kg CO_2 / kg LW at the farm gate, not taking into account the multiple output system presented in Phong *et al.* (2011). Poultry systems do not have enteric emissions, normally the impact of the previous generation is considered negligible (breeding stock produce a lot of chicks), and the feed conversion ratios (FCR) are relatively low in comparison to other land animals. There is an agreement that feed production is the most important life cycle stage in the GHG emissions of poultry meat systems with example relative contributions of 82% (Pelletier, 2008) and 83% (Cederberg *et al.*, 2009a). Manure management can provide negative emissions in some studies, as in Pelletier (2008), because of the avoidance of inorganic fertiliser production. Two studies on poultry meat that are related (Williams *et al.*, 2006; Leinonen *et al.*, 2012) included the effect of the breeding stock.

It should be noted that transport of feeds even in cases where studies focus solely on feed production and delivery is not a major contributor in comparison to agricultural emissions (Dalgaard *et al.*, 2008). The transport of animals does not seem to be of importance in comparison to agricultural emissions either (Dalgaard *et al.*, 2007). In fact even when the transport of meat produced in Brazil to Europe has been included, it has been shown that transport was not an important contributor in comparison to other sources in livestock production systems (Cederberg *et al.*, 2009b). In general, transport does not seem an important relative contributor in meat production systems, as these systems are mostly characterised by emissions associated with agricultural processes (and/or land transformation).

3.6 Discussion

Every study or review that has investigated the GHG emissions from a range of animal production systems has found that the meat produced from birds has a similar or slightly lower GHG emissions intensity than those of pig systems, and that both are considerably lower than those of ruminant systems (Williams *et al.*, 2006; Weidema *et al.*, 2008b; Cederberg *et al.*, 2009a; de Vries and de Boer, 2010; Dyer *et al.*, 2010; Lesschen *et al.*, 2011). The reasons for the difference are quite obvious, birds are very efficient protein producers and accumulators (as evidenced by a relatively low FCR), manure emissions are not as high as for other animal systems, the impact from the previous generation is negligible, and emissions from enteric fermentation are insignificant. Pigs are also monogastric animals but are less efficient than birds, emissions from enteric fermentation are not negligible, and emissions from manure management can be important. In addition the effect of the previous generation is important in mammals. At the higher extreme in GHG emission intensity of meat production systems are ruminant systems. Ruminants have relatively high FCR, the impact from progeny is critical (cows only have one calf per year), emissions from manure management is of consideration and emissions from enteric fermentation are relatively high. There is great disparity in the GHG results for beef production because of variations between studies in all these important factors. It should be noted that manure management, depending on the LCA modelling choices can provide emission credits through the avoidance of inorganic fertilisers or fossil energy production.

Figure 23 presents some of the GHG emissions results in this review based on LW. Weidema *et al.* (2008b) is not included because it is a cradle-to-grave study, and thus not comparable with the rest of the studies illustrated in Figure 23. Beauchemin *et al.* (2011) is not included as it is the same base case in Beauchemin *et al.* (2010) which is included. Phetteplace *et al.* (2001) and Veysset *et al.* (2010) have not been included as they presented different types of animal husbandry operations (e.g. calf-to-weanling, calf-to-beef systems) and not necessarily average intensity or CW or LW. Leinonen *et al.* (2012) was not included as it is a related study to Williams *et al.* (2006) and results for the latter based on LW have been provided (pers. comm. Table A-4 in Appendix A) and are here assumed to be valid for both. Figure 23 only includes average figures for Brazil and does not include results for newly deforested land that are detailed in Cederberg *et al.* (2011). Phong *et al.* (2011) is not included because it is a very special multiple system (agriculture-aquaculture system). Edward-Jones *et al.* (2009) is not included as it is a case for multiple species farms (beef and lamb). Cases that include the opportunity cost of land in Nguyen *et al.* (2010a) and land transformation in Nguyen *et al.* (2010b) are not included. Dalgaard *et al.* (2007) has not been included as it uses an extreme consequential approach. For studies with results based only on CW, the carcass yields were assumed to be 58%, 47%, 75% and 70%, for beef, lamb, pig, and chicken systems, respectively. LCA has been performed also for a variety of aquaculture systems (Henriksson *et al.*, 2011). Although salmonid farming is not currently a major meat producing sector in comparison to land based systems, aquaculture is the animal food production sector with the highest growth rate. Global farmed salmon, trout and smelts production increased from 1.4 to 2.3 million tonnes between 1999 and 2008 (FAO, 2008a); a 64% increase. Therefore some results in the literature for farmed salmonids (salmon and trout) (Aubin *et al.*, 2009; Ayer and Tyedmers, 2009; Pelletier *et al.*, 2009) have been included in Figure 23. The result for the recirculating salmon aquaculture system in Ayer and Tyedmers (2009) is not included as it is a niche technology. All these studies on salmonid systems used LW as functional unit (the actual outcome of the farm).

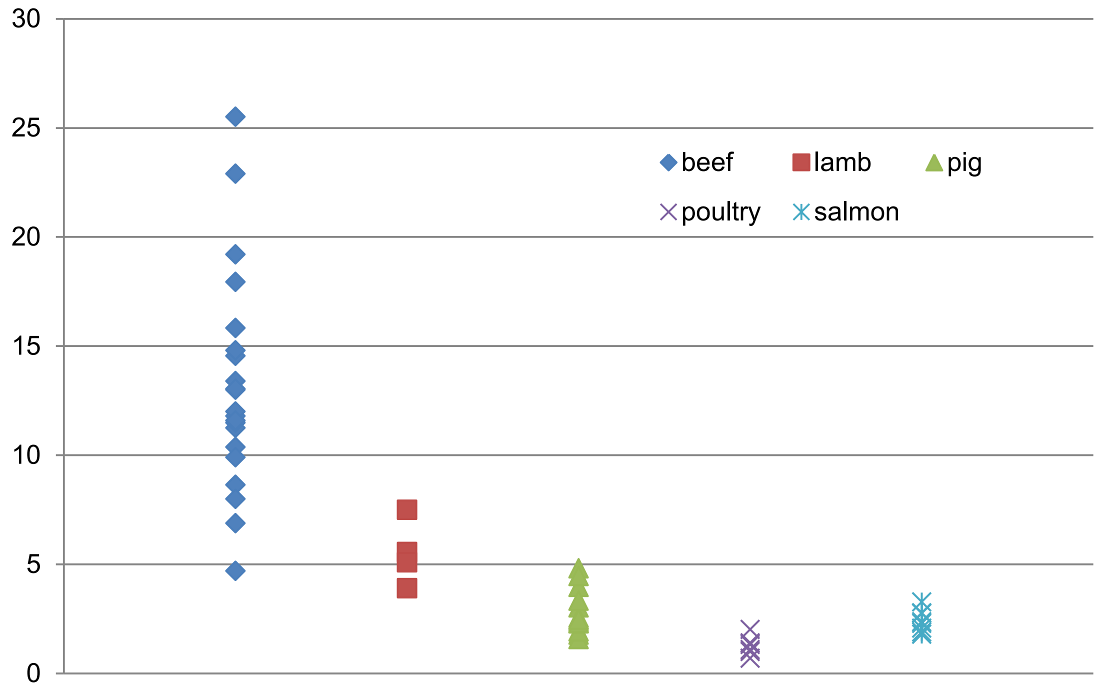

Figure 23 Greenhouse gas emissions of beef, lamb, pig, poultry and salmonid production systems (kg CO_2e/ kg live weight)

Results in Figure 23 range from 4.7 to 25.5, 3.9 to 7.5, 1.6 to 4.8, 0.7 to 2, and 1.8 to 3.3 kg CO_2e / kg LW for beef, lamb, pig, poultry and salmonid systems respectively. Ruminant systems (beef and lamb) present the greatest range of results (Figure 23), possibly because ruminant systems have many important processes affecting their GHG emissions (see above). Furthermore beef system studies are the most numerous, taking into account that ruminant system results depend on many factors, and therefore there is more possibility for different results. The pig, poultry and salmonid systems present similar results (at least when compared to ruminants systems). Of the currently commercially available meat production systems, poultry systems appear to be associated with the lowest GHG emission intensity.

Figure 24 presents the studies based on edible meat produced, calculated as 90% of the CW as in Lesschen *et al.* (2011). Where the results of studies have been presented based on the edible portion it has been used directly in this form. Carcass yield factor in salmonid systems has been assumed as 50 %, similar to processing of oily fish in Nielsen *et al.* (2003). Figure 24 also includes results for in-vitro production of cultured meat (Tuomisto and de Mattos, 2011).

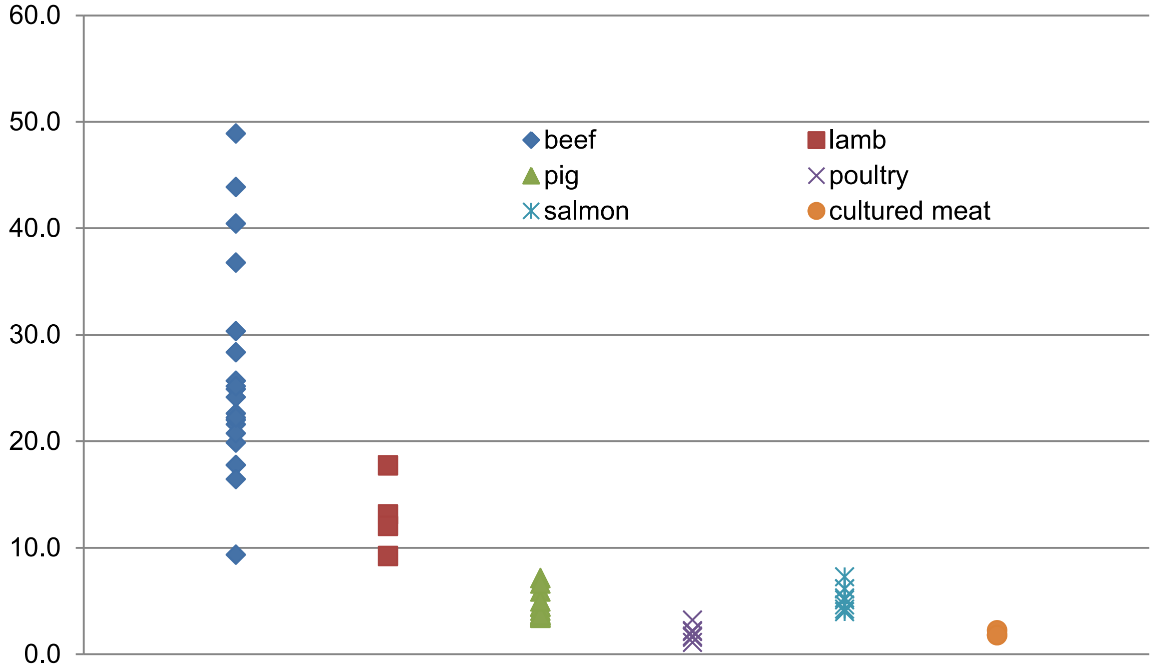

Figure 24 Greenhouse gas emissions of beef, lamb, pig, poultry, salmon and cultured meat production systems (kg CO_2e/ kg edible meat)

Although results from different studies are not necessarily comparable, it seems that the meat production systems associated with the lowest GHG emission intensities are poultry and cultured meat systems. Pig and salmon systems have similar GHG intensity when compared to ruminant systems. Cultured meat production systems are only at experimental level today and therefore are not currently commercially produced.

An evaluation of improvement options on beef systems has been presented in Beauchemin *et al.* (2011) and in general for land based animal production in Weidema *et al.* (2008b). The former focuses on on-farm strategies for 100% beef systems and the latter focuses on overall improvements in the livestock system. Strategies including feeding oilseeds, improving forage quality and increasing the number of calves weaned show great potential for mitigation in beef systems. Beef from milk-beef systems have a lower GHG intensity than 100 % beef systems, and therefore the improvement options in beef and dairy have to be evaluated together. A global perspective for mitigation strategies in animal production has been discussed in de Boer *et al.* (2011). Mitigation strategies in monogastric land based meat production systems (pigs and birds) are more related to improvement in feed conversion and mitigation of emissions from the feed production system and manure emission.

Being an important source of GHG's, it is important to work towards the mitigation of emissions from every meat production system. However, it is also important to realise that an important step towards emission mitigation would be a shift from high GHG emission intensive systems (as ruminants) to more efficient systems. It is important to notice that the present review only focuses on Climate Change impact. Other environmental impacts should be taken into account when the environmental relevance of a system is to be described or compared. Furthermore, other aspects of sustainability beyond the environmental dimension should also be considered.

The present work is not an attempt of providing a detailed review of GHG emissions from animal production systems, rather a broad perspective on differences between studies. It is however obvious that research into the GHG emissions from meat production systems is in a mature stage regarding animal production; however there is need of further work into evaluating consequences, principally associated with current and potential land transformation. There is also further work needed into accounting for GHG emissions arising during slaughtering, and in particular ways of accounting for animal by-products processing.

3.7 Conclusion

In general, GHG emission results from land-based meat production systems vary between different studies. Calculation methods of economic flows varies with some studies having a national intensity approach (top-down), and other studies having a more traditional process based LCA (bottom-up) approach. Some studies used both and this is called the "hybrid" approach to calculation. The calculation of natural flows is normally based on standard methodologies, emission factors or databases. Whole farm simulation approaches are used frequently. It seems that there is plenty of literature already for ruminant products; however studies for monogastric systems are not as common.

Critical methodological issues found were: co-product handling when studying milk-beef systems and using a consequential approach in particular in land use in animal and feed production. Geographical differences are important and may be related to: animal management, feed ingredients sourcing, and in general environmental burden of the inputs and outputs (e.g. the GHG emission intensity of energy systems vary from country to country).

It is clear that studies on animal production are focused on the agricultural aspects of the production chain which are probably the most important. Results for cradle-to-farm-gate are expressed either on CW or LW. Using CW seems more useful as it is closer to the sealable part; however LW seems more ideal from a traditional LCA perspective as the actual outcome of the farm are live animals.

Chapter 4

Greenhouse gas life cycle assessment of products arising from the rendering of mammalian animal by-products in the UK

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4 Greenhouse gas life cycle assessment of products arising from the rendering of mammalian animal by-products in the UK

4.1 Abstract

Animal by-products (ABP) are unavoidable by-products of meat production that are categorized under EU legislation into category 1, 2, and 3 materials, which are normally treated by rendering. Rendering is a thermal process that produces rendered fat and protein. Heat is provided from the combustion of natural gas and self-produced rendered fat. The main objectives of the study were (i) to assess energy intensity in the UK rendering industry, and (ii) to quantify the greenhouse gas emissions associated with the production of mammalian rendered products using life cycle assessment.

Thermal energy requirements were 2646 and 1357 kJ/kg, whereas electricity requirements were 260 and 375 kJ/kg for category 1 and 3 ABP respectively. Fossil CO_2 emissions were −0.77 and 0.15 kg CO_2e/kg category 1 and 3 mammalian rendered fat respectively and 0.15 kg CO_2e/kg processed animal protein. These were low relative to vegetable products such as palm oil and soya bean meal because (i) ABP were considered wastes that do not incur the environmental burden of their production, and (ii) the rendering process produces biofuels that can be used to generate energy that can be used to offset the use of fossil fuels in other systems.

4.2 Introduction

Animal by-products (ABP) are secondary products of the animal production and meat industries that can account for up to 50% of the live weight of an animal. ABP are classified by European legislation into three categories (EC, 2009). Category 1 material includes animals or body parts infected with, or suspected of being infected with, a transmissible spongiform encephalopathy and includes skull, brain, tonsils, spinal cord, and intestines of bovine animals, entire head and vertebral column. Category 1 materials must be disposed of by combustion, either directly or following rendering. They can be used as biofuels and their energy content recovered. Category 2 material mainly includes fallen stock, manure and digestive tract content. Category 2 materials can be used for composting, biogas production, or following rendering they can be used to manufacture fertilizer and in the oleochemical industry. Category 3 materials include parts of slaughtered animals fit for human consumption, or unfit but with no transmissible diseases (including hides, skins, horns, bristles and feathers, non-ruminant blood, raw milk from healthy animals, and food of animal origin which is no longer able to be consumed by humans). Category 3 materials can be used in pet food manufacturing. When different categories of material are mixed together they are downgraded to the lower number category (EC, 2009). For example, category 1 and 2 materials are usually mixed together in the UK and processed together as category 1 material. This classification establishes a marked difference between different categories of ABP as category 1 material has to be destroyed. In contrast, category 3 material is potentially a valuable commodity. However, disposal of mammalian ABP from all categories represent a cost to meat producers in the UK.

Woodgate and van der Veen (2004) define rendering as "to render open or split • by heat processing • raw material into a solid protein meal and a liquid". Dry rendering systems are the most common (Anderson, 2006). In this process, the raw material (ABP) is ground and passed through a disk dryer/cooker. The vapour from the cooker is typically taken to a condenser. The process condensate is then sent to wastewater treatment and noncondensable gases are treated to destroy odours. Alternatively in some systems the vapour is sent to a thermal oxidizer and the remaining gas is released to the atmosphere. The dry material is directed to a press where the liquid (fat) is separated from the solid (proteinaceous material). The solid is ground to obtain protein meal and the liquid is directed to a filtration system to obtain rendered fat (tallow in the case of beef fat). Within the EU, category 1 protein meal is referred to as meat-and-bone-meal (MBM), and category 3 protein meal is referred to as processed-animal-protein (PAP).

Life cycle assessment (LCA) is a mature ISO-standardized tool for evaluating the potential environmental impacts from products or services throughout their life cycle (ISO, 2006a; b) where the life cycle is defined as the series of interlinked stages required to produce, use and dispose of a product. However, LCA methodology may also be used to evaluate only parts of the product life cycle. The choice of environmental impact categories (climate change, eutrophication etc.) depends on the objectives of the study. LCA has been used extensively to evaluate the environmental impacts of producing different fats and protein meals (Dalgaard *et al.*, 2008; Reijnders and Huijbregts, 2008; Schmidt, 2010).

To date the UK is the only country to have established a legally binding long-term framework to reduce greenhouse gas (GHG) emissions. The UK Climate Change Act 2008 (UK, 2008a) sets ambitious targets to reduce GHG emissions to 80% below 1990 levels by 2050. Industry has a major role in the attainment of these targets. Consequently it is important that industries, such as the rendering industry, quantify their GHG emissions to enable sensible carbon management strategies to be developed. The GHG emissions associated with the rendering process are due to energy use, wastewater treatment and the production of chemicals that are used in the process. The use of rendered fat as a fuel instead of fossil fuels in the rendering process could have critical implications regarding GHG emissions. As rendered fat is produced from animal material, the CO_2 emissions produced from its combustion are biogenic, and hence can be considered carbon neutral. This is because biogenic CO_2 is associated with a short carbon cycle (Solomon *et al.*, 2007b), in which carbon absorbed from the atmosphere by plants during photosynthesis is returned to the atmosphere when rendered fat is burnt to produce energy. Emissions from the combustion of fossil fuels lead to a net gain in atmospheric CO_2. To date, there has been no study that analyses the effect on CO_2 emissions of using varying proportions of fossil fuel and rendered fat for the production of thermal energy in the rendering process.

The objectives of the study were (i) to assess the energy intensity of the UK rendering industry, and (ii) to quantify the GHG emissions associated with rendered products derived from mammalian by-products using LCA methodology. The effect of the fuel type used to produce process heat (i.e.,natural gas or rendered fat) on GHG emissions was also investigated.

4.3 Materials and methods

4.3.1 System description, boundaries, and co-product handling

Three functional units were used: 1 kg category 1 mammalian rendered fat (MRF), 1 kg category 3 MRF and 1 kg PAP (at the rendering plant gate). Two product systems were defined: S1 for category 1 MRF and S2 for category 3 rendered products (Figure 25). As the disposal of mammalian (mainly ruminant material) ABP represents a cost to meat producers and economic flows with negative value are treated as wastes in LCA methodology (Guinee *et al.*, 2004; Guinee *et al.*, 2009), the production of ABP, including animal production and processing, were not included in the system boundaries.

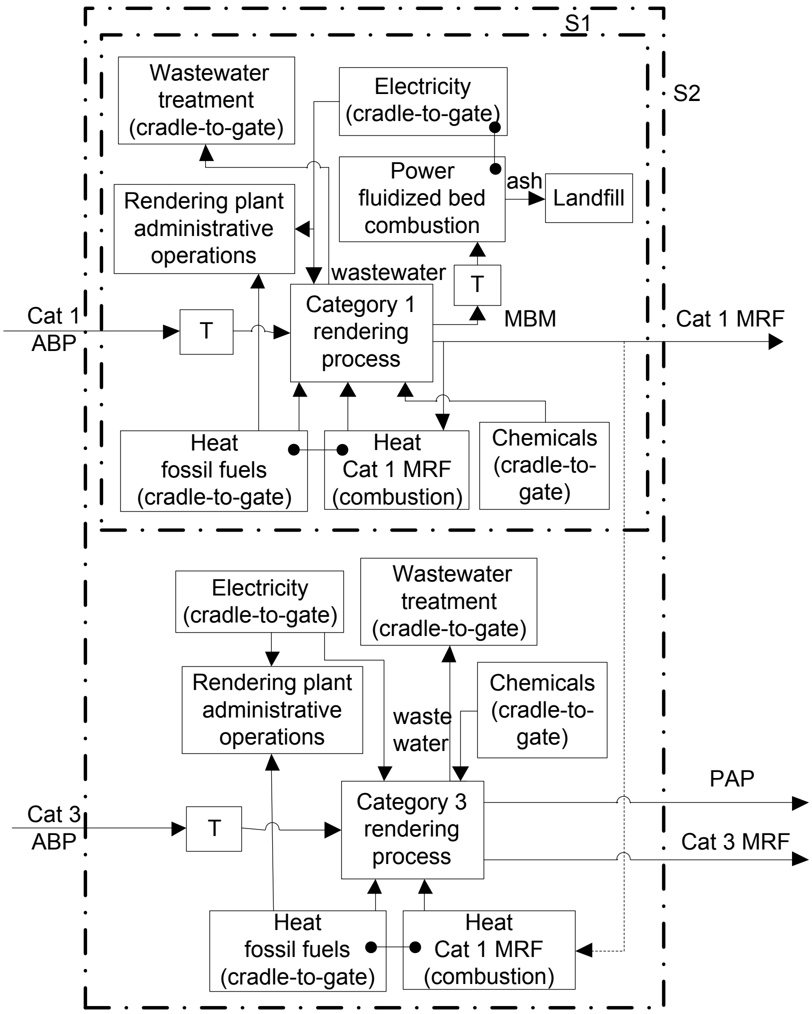

Figure 25 System boundaries for the production of category 1 and 3 mammalian rendered fat (MRF), and processed animal protein (PAP) by the rendering process (ABP: animal by-product, Cat: category, T: Transport, MBM: meat and bone meal)

The product systems are described as follows:

S1: The product system in which category 1 and 2 ABP are converted to category 1 MRF. The system was expanded to replace average British electricity production with electricity produced from the combustion of MBM in Fluidised Bed Combustion (FBC) power plants. FBC power plants are the most common disposal option for MBM in the UK (personal communication with Stephen L. Woodgate, FABRA). A test for avoidance of alternative electricity technologies has been included in the Sensitivity Analysis (Supporting Information). The use of category 1 MRF as fuel in the rendering process can be considered a recycling process (See the Fuel Scenarios section).

S2: The product system in which category 3 ABP are converted to category 3 MRF and PAP. S2 includes S1, as category 1 MRF is used as a fuel in the category 3 rendering process. Co-product handling of GHG emissions between category 3 MRF and PAP was completed using mass and economic allocation and system expansion. The mass allocation factors employed were based on yield, where yield was defined as the mass of ABP that was converted to a rendered product (i.e., rendered fat or protein meal). The mass allocation factors used were 57.7% for category 3 MRF and 42.3% for PAP. Economic allocation was performed according to the average prices of rendered products between September 2007 and January 2010 provided by a rendering company. During the period studied the price ratio of rendered products varied (Figure 26). The economic allocation factors used were 77.7% for category 3 MRF and 22.3% for PAP. System expansion assumed that the main product was category 3 MRF because the greatest revenue for a rendering system comes from the production of MRF. As the current most valuable use of PAP is as an ingredient for pet food manufacturing, in the system modelled, the coproduction of PAP avoids the production of similar protein meals. Soybean meal was chosen based on its similar total protein content: the total crude protein contents of PAP and soybean meal being 538 and 516 g/kg dry matter respectively (Sellier, 2003).

Systems S1 and S2 also include ABP transport to the rendering plant, MBM transport from the rendering plant to the FBC power plant, electricity, water, wastewater treatment and the production of chemicals used in the rendering process (Figure 25). Infrastructure was not included and a rationale for its exclusion is provided in the Sensitivity Analysis.

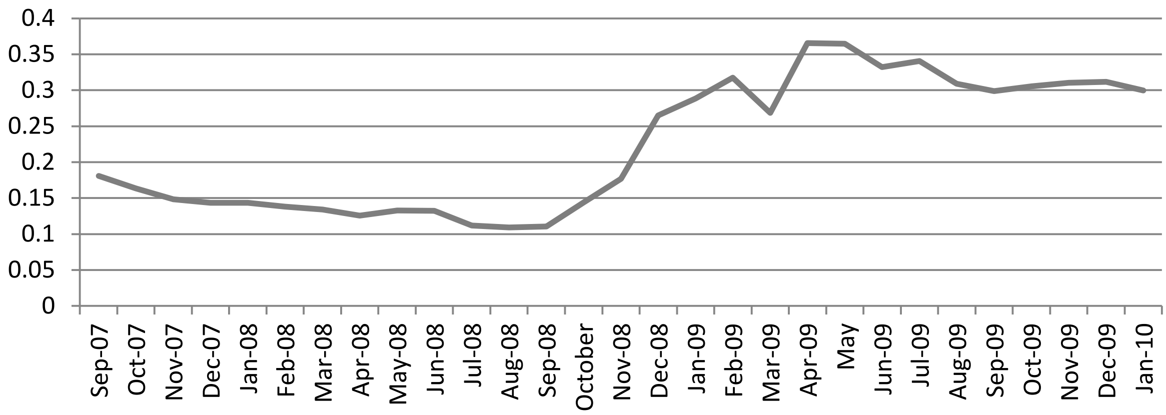

Figure 26 Price ratio of category processed animal protein (petfood grade) to category 3 mammalian rendered fat (grade 2) on a mass basis between September 2007 and January 2010 (provided by an anonymous UK company)

4.3.2 Fuel scenarios

The UK rendering industry uses both natural gas and category 1 MRF as fuel to produce heat for the rendering process. The proportion of each fuel used varies on an annual basis. The impact of using different proportions of natural gas and category 1 MRF on the life cycle GHG emissions associated with the functional units was investigated. The system was tested assuming that 0%, 25%, 50%, 75%, and 100% of the thermal energy was derived from natural gas, with the remainder being derived from MRF. When category 1 MRF is used as a fuel within the system, more category 1 MRF is produced than the actual functional unit depending on the level of substitution. When category 3 rendered products were produced using category 1 MRF as a fuel, both the category 3 rendering process and the category 1 rendering process were modeled using the same percentage (e.g., for the production of category 3 MRF that was rendered using 25% natural gas, the remaining 75% came from category 1 MRF that had been rendered using 25% natural gas and 75% category 1 MRF).

4.3.3 Data and calculation

The study used primary and secondary data sources. Category 1 and 2 ABP are usually mixed and processed together as category 1 material. Total processing of category 1 and 2 ABP in the UK between 2006 and 2007 was approximately 1 150 000 tonnes per annum, while the processing of category 3 ABP was approximately 950 000 tonnes per annum (pers. comm. with SL Woodgate).

Primary data for the years 2006, 2007, and 2008 was collected through direct contact with UK rendering plants. Specific primary data collected was:

1. The yield of rendered fat and protein meal produced by category 1 and 3 rendering plants. Data was obtained from seven rendering plants (five category 1 plants and two category 3 plants), which processed between 30% and 40% of UK ABP.

2. The annual amount of ABP processed. Electricity consumed and fuels used was obtained through direct contact with five rendering plants, which processed approximately 30% of category 1 ABP and 10% of category 3 ABPs in the UK.

3. The annual amount of water and chemicals used was collected from 4 UK rendering plants for the years 2006, 2007, and 2008. These plants processed approximately 20% of UK ABP.

4. The annual amount of wastewater produced was collected from three rendering plants, which processed approximately 15% of UK ABP.

5. The annual amount of ash produced from the combustion of a certain amount of feedstock in a FBC system was collected from 1 UK FBC power plant.

Simapro 7 (PRe Consultants, 2011) was used for system modelling and calculation of results. Climate change was assessed using the Greenhouse Gas Protocol 1.00 (The Greenhouse Gas Protocol, 2010) impact assessment method, which makes a distinction between fossil and biogenic carbon, and uses the climate change characterization factors with a time frame of 100 years reported by IPCC (Solomon *et al.*, 2007b). Data sources for economic flows and life cycle inventory are presented in the Appendix B.

4.4 Results

4.4.1 Rendered product yields

The current study collected data from rendering plants that processed between 30% and 40% of UK ABP between 2006 and 2008, and thus the yields presented can be considered a robust average. Table 14 presents the average economic flows for the rendering of 1 kg of category 1 ABP and 1 kg of category 3 ABP, respectively. Yields of 0.27 kg MBM/kg ABP processed and 0.13 kg MRF/kg ABP processed were determined for category 1 material. Yields of 0.33 kg PAP/kg ABP processed and 0.24 kg MRF/kg ABP processed were determined for category 3 material.

Table 14 Economic flows for the rendering of 1 kg Category 1 and 3 mammalian ABP in the UK

Product flow	Units	Category 1 rendering			Category 3 rendering		
		Amount	min	max	Amount	min	max
MBM produced	kg	0.27	0.24	0.32	n.a.	n.a.	n.a.
PAP produced	kg	n.a.	n.a.	n.a.	0.33	0.26	0.38
Category 1 MRF	kg	0.13	0.09	0.17	n.a.	n.a.	n.a.
Category 3 MRF	kg	n.a.	n.a.	n.a.	0.24	0.18	0.29
Heat (as energy content in fuel)	kJ	2646	2218	3075	1357	1333	1394
Electricity	kJ	260	154	333	375	361	383
Water use	m^3	0.00179	0.00104	0.00286	0.00165	0.00148	0.00181
Wastewater[a]	m^3	0.00174	0.00072	0.00326	0.00174	0.00072	0.00326
Chemicals use							
Sodium Hypochlorite	kg	3×10^{-6}	2×10^{-6}	4×10^{-6}	3×10^{-7}	8×10^{-8}	8×10^{-7}
Sodium Hydroxide (Caustic)	kg	6×10^{-7}	2×10^{-7}	1×10^{-6}	1×10^{-7}	3×10^{-8}	1×10^{-7}
Sulphuric Acid	kg	4×10^{-7}	5×10^{-8}	9×10^{-7}	1×10^{-7}	6×10^{-8}	2×10^{-7}
Various boiler, cooling tower and cleaning chemicals	kg	9×10^{-7}			1×10^{-6}		

min (minimum value)
max (maximum value)
MBM (meat and bone meal)
PAP (processed animal protein)
MRF (mammalian rendered fat)
n.a. : not applicable
[a]wastewater is taken from category 1 rendering plants only and applied to both category 1 and 3 rendering plants

4.4.2 Energy intensity of the UK rendering process

Thermal energy requirements for the rendering of category 1 and 3 ABP were 2646 and 1357 kJ/kg ABP processed respectively (Table 14). This energy was produced from the combustion of category 1 MRF and natural gas, and was calculated by adding the amount of energy contained in natural gas to the amount of energy contained in MRF. The average does not take into account the combustion efficiency of the different fuels. The amount of thermal energy needed to treat category 1 was higher than that required to treat category 3 ABP. Electricity consumption in the current study was 260 and 375 kJ/kg ABP processed for category 1 and 3 material, respectively (Table 14).

4.4.3 Category 1 MRF life cycle GHG emissions

The life cycle CO_2 emissions associated with category 1 MRF were dependent on the proportion of natural gas and MRF used. Fossil CO_2 emissions ranged from −1.61 to 0.40 kg CO_2e/kg for category 1 MRF (Figure 27), increasing as the percentage of natural gas increased.

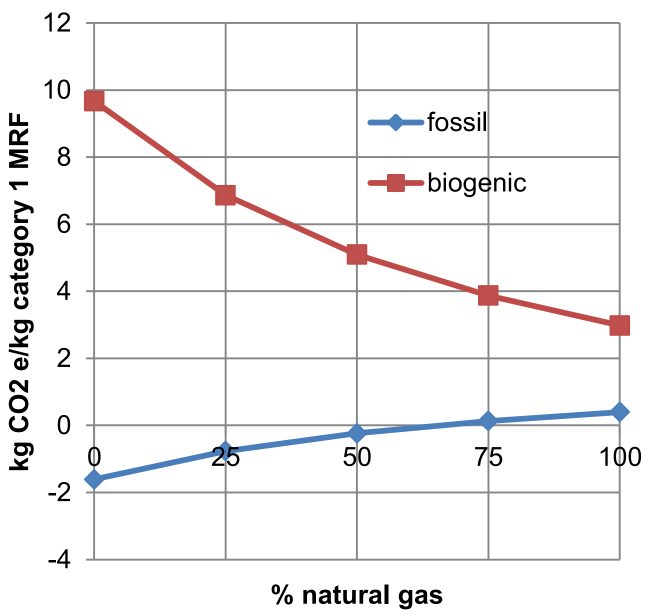

Figure 27 Fossil and biogenic CO_2 emissions for 1 kg of category 1 mammalian rendered fat (MRF) based on the percentage of natural gas used as fuel for the rendering process (the rest being provided by category 1 MRF as fuel)

Negative fossil CO_2 emissions occurred when the percentage of natural gas used was between 0% and 70% (Figure 27). Biogenic CO_2 emissions decreased with increasing natural gas use, ranging from approximately 3 to 10 kg CO_2e/kg category 1 MRF. This was because when more natural gas was used, less category 1 MRF was used. Biogenic CO_2 emissions are associated with the combustion of category 1 MRF as thermal fuel for the rendering process and MBM to produce process electricity in FBC plants.

4.4.4 Category 3 MRF life cycle GHG emissions

The life cycle CO_2 emissions for category 3 MRF varied depending on the percentage of MRF used as fuel and the allocation approach employed (Figure 28). Fossil CO_2 emissions increased with increasing natural gas use for each allocation approach employed, ranging between −0.11 and 0.54 kg CO_2e/kg category 3 MRF. The lowest figure for fossil CO_2 emissions was obtained for each allocation approach when the amount of natural gas used was 0%. The CO_2 emissions varied depending on the approach used to allocate emissions between category 3 MRF and PAP. Economic allocation resulted in the highest figures for fossil CO_2 emissions, with emissions ranging between 0.09 and 0.54 kg CO_2e/kg category 3 MRF. Mass allocation resulted in fossil CO_2 emissions ranging from 0.05 to 0.29 kg CO_2e/kg category 3 MRF (Figure 28). For system expansion, negative fossil CO_2 emissions were calculated when the percentage of natural gas used in the rendering process was low (less than 12%: Figure 28). When the proportion of natural gas used was 0%, this approach provided the lowest result (−0.11 kg CO_2e/kg category 3 MRF). This is because of the double effect of using category 1 MRF produced with 0% natural gas as a fuel and the avoidance of soybean meal production.

Biogenic CO_2 emissions ranged between 0.01 and 1.94 kg CO_2e/kg category 3 MRF depending on the allocation approach employed and the percentage of natural gas used (Figure 28). For each allocation approach used, biogenic CO_2 emissions increased with decreasing natural gas use because the combustion of tallow and MBM contributes to biogenic CO_2 emissions. System expansion resulted in the highest biogenic CO2 emissions.

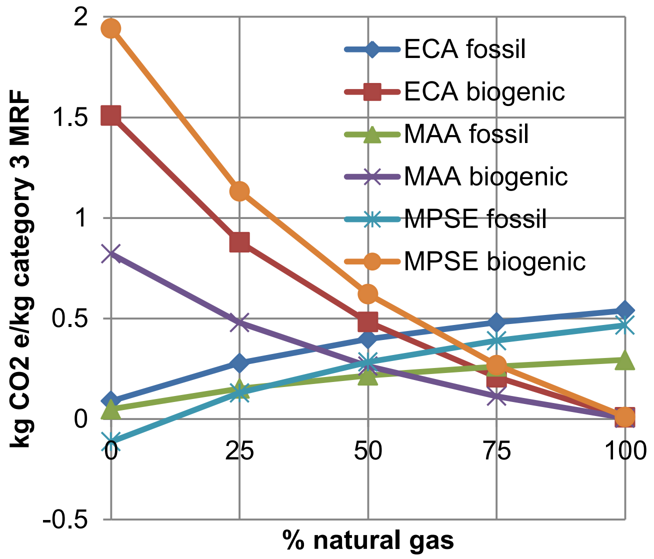

Figure 28 Fossil and biogenic CO_2 emissions for 1 kg of category 3 mammalian rendered fat (MRF) based on the percentage of natural gas used as fuel for the rendering process (the rest being provided by category 1 MRF as fuel) for each co-product handling approach (ECA: economic allocation, MAA: mass allocation, MPSE: system expansion).

4.4.5 PAP life cycle GHG emissions

The CO_2 emissions associated with PAP (Figure 29) were similar to those for category 3 MRF, as the systems were the same and only the allocation factors used were different.

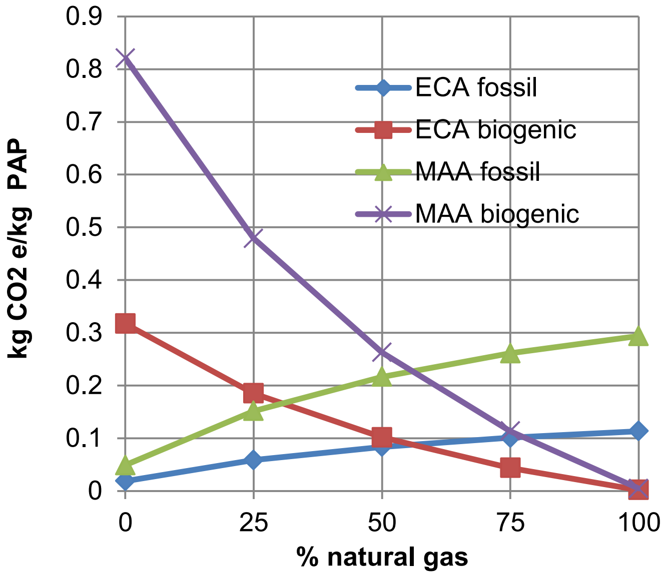

Figure 29 Fossil and biogenic CO_2 emissions for 1 kg of processed animal protein (PAP) based on the percentage of natural gas used as fuel for the rendering process (the rest being provided by category 1 tallow as fuel) for each co-product handling approach (ECA: economic allocation, MAA: mass allocation).

There are no results for system expansion for PAP because this is the co-product that avoids production of soybean meal and consequently category 3 tallow takes the credits for this. Fossil CO_2 emissions from the PAP system ranged from 0.00 to 0.29 kg CO_2e/kg PAP depending on the allocation approach and percentage of natural gas used in the system (Figure 29). Contrary to category 3 MRF, the highest CO_2 emissions occurred when mass allocation was used, and the lowest when the economic allocation approach was used. Fossil CO_2 emissions were equal to those of category 3 MRF when mass allocation was used (Figure 29).

4.5 Sensitivity analysis

A sensitivity analysis was performed to test some of the assumptions made in the product system modelled. The base case results are set as the fossil CO_2 emissions for category 1 and 3 MRF with 25% of the thermal energy requirement being provided from natural gas and 75% from category 1 MRF. Allocation of CO_2 emissions between category 3 MRF and PAP was based on mass. Base case results were -0.77, 0.15 and 0.15 kg CO_2e/kg category 1 MRF, category 3 MRF and PAP respectively.

4.5.1 Thermal energy requirement of the category 3 system

Both the yield of rendered products and the thermal energy requirement of rendering vary between literature references. Thermal energy is required to evaporate water and produce dry rendered products. In the current study, a difference in the thermal heat required to render category 1 and 3 ABP was obtained. The thermal energy required to render category 1 animal ABP being higher than that reported in the literature. A test was performed to evaluate the impact of applying the thermal energy required to process category 1 ABPs to category 3 ABPs, but keeping the yield of rendered products the same. CO_2 emissions from both the category 1 and 3 systems were only marginally affected by this change. As more thermal energy is required, more category 1 MRF is used in the category 3 system. As a consequence, more category 1 MBM is co-produced that is disposed of by combustion in FBC power plants. As a result the production of more British electricity is avoided. This effect is particular for the UK rendering industry and the results would be different if the system used less category 1 MRF, and more natural gas fuel to generate thermal energy. It would also be different if a less carbon intensive source of electricity was avoided by the combustion of MBM.

4.5.2 Source of electricity avoided by combustion of MBM

In the base case system modelled production of British electricity is avoided by the combustion of MBM in FBC plant. British electricity is highly coal based and therefore very carbon intensive. Two test were performed: 1) Assuming that the electricity avoided was produced from natural gas and 2) Assuming that the electricity avoided had a very low carbon intensity (average Norway electricity).

If the electricity avoided was produced from natural gas, fossil CO_2 emissions increased to -0.16, 0.18 and 0.18 kg CO_2e/kg category 1 MRF, category 3 MRF and PAP respectively. The result is important for category 1 MRF. In this scenario emissions from the category 1 system are still negative, as in the base case. Electricity production from natural gas is marginal in the UK, and results for electricity production avoided, either using British electricity or natural gas are in reasonable agreement.

If the electricity avoided was produced from low carbon sources (Norway electricity), fossil CO_2 emissions increased to 1.26, 0.25 and 0.25 for category 1 MRF and category 3 MRF and PAP respectively. In this scenario emissions from the category 1 system would be positive and higher than those from the category 3 system. However, in the UK this scenario does not currently apply as electricity is highly carbon based.

This test illustrates that the results obtained are dependent on the carbon intensity of electricity production where the system is modelled. The base case is relevant to the UK, but should be treated with caution if applied to systems outside the UK.

4.5.3 Inclusion of infrastructure in the system boundaries

In the base case infrastructure has not been included in the system boundaries. The main capital good used in the system is the rendering plant (buildings and equipment). A test was performed assuming that the life-span of a rendering plant is 20 years with an average throughput of 100,000 tonnes of ABP/year. If the life cycle inventory for a similar sized chemical plant in Germany was included in the system boundaries CO_2 emissions would be -0.77, 0.15 and 0.15 for category 1 MRF, category 3 MRF and PAP respectively. The inclusion of infrastructure in the system modelled would have no impact on the results obtained.

4.6 Discussion and conclusion

4.6.1 Rendered product yields and energy intensity

Data on the quantity of ABP processed, energy use and rendered fat and protein meal yields was collected from UK rendering plants which processed 30−40% of UK ABP between 2006 and 2008.

Consequently, the yields obtained can be considered representative of the UK rendering industry. Both MRF and protein meal yields were higher for category 3 than category 1 materials, suggesting that category 3 ABP contained less water. The yields of tallow and protein meal reported by Lopez *et al.* (2010) for the US rendering industry were 0.28 and 0.23 kg/kg ABP processed, respectively. Differences in rendered product yields between the US and UK can be explained by differences in the composition of ABP processed. In the US, ABP are not classified in the same way as they are under EU legislation, with no differentiation between category 1, 2, and 3 materials.

In the current study, the amount of thermal energy used to process category 1 ABP was higher than that used for category 3 ABP. However, there was little difference between category 1 and 3 rendering plants in electricity use. As thermal energy is primarily used to generate process heat to evaporate water, this again suggests that category 1 ABP contained a higher proportion of water. Electricity is primarily used for motion and process control systems, which do not differ greatly between category 1 and category 3 rendering plants. Lopez *et al.* (2010) reported average thermal energy and electricity requirements of 2113 and 292 kJ/kg ABP processed respectively, which are midway between the values for category 1 and 3 ABP obtained in the current study. As stated above differences in energy use probably reflect differences in the composition of ABP processed in the US and UK.

The Ecoinvent database (Ecoinvent Centre, 2010) provides an inventory for tallow production for a rendering plant in Switzerland and provides both thermal energy and electricity values for the production of 1 kg of tallow. If these values are applied to the MRF yields obtained in the current study (0.13 and 0.24 kg MRF/kg ABP processed for category 1 and 3 material respectively) the thermal energy requirement for category 1 and 3 would be 1090 and 2000 kJ/kg ABP processed, respectively. Similarly, the electricity requirement would be 82 and 151 kJ/kg ABP processed, respectively. In the current study, the thermal energy requirements was higher for category 1 and lower for category 3 material than calculated using the Ecoinvent database. Similarly, the electrical energy requirement for both category 1 and 3 materials was higher than calculated using the Ecoinvent database.

Ramírez *et al.* (2006) reported a primary energy requirement for rendering of 1625 kJ/kg ABP processed. Primary energy represents energy embodied in natural resources such as coal, before it is converted into usable energy such as electricity. Consequently, calculation of primary energy includes the inefficiency of electricity production, using an efficiency of 40%.18 If this approach is applied to data collected in the current study, the primary energy requirements would be 3296 and 2293 kJ/kg ABP processed for category 1 and 3 materials respectively. These values are higher than those reported by Ramírez *et al.* (2006). Overall, the thermal and electrical energy requirements of rendering depends on the composition of ABP processed, with more thermal energy being required to process material with higher water content. The values obtained in the current study are in the same order of magnitude and consistent with those obtained in other studies.

4.6.2 Greenhouse gas emissions from rendered products

Under EU legislation (EC, 2009) secondary products of animal production and meat processing are classified as by-products, which potentially could be considered as commodities with economic value. However, In the UK the disposal of aggregated mammalian ABP of all categories represents a cost to the producer. Within LCA methodology, economic flows with negative economic value are considered to be wastes (Guinee *et al.*, 2004). Therefore, the environmental burden associated with their production should be allocated to other products such as meat, and hides and skins, which have a positive economic value. In the UK, mammalian ABP should not carry any of the environmental burdens associated with their production. This is not necessarily the case in other countries, or for ABP derived from other species. For example, in Australia ruminant ABP may have a positive economic value (Beer *et al.*, 2007). Similarly, in the UK category 3 ABP derived from poultry have a positive economic value. In which case, the environmental burden associated with their production and processing could be allocated accordingly.

In the current study, GHG emissions are reported on a CO_2 equivalent basis (CO_2e). It should be noted that the combustion of natural gas to produce thermal energy produces both CO_2 and a small proportion of CH_4 as GHGs. However, the proportion of CH_4 is negligible in comparison to CO_2 emission. Consequently, fossil CO_2 emissions can be considered to be almost equal to fossil GHG emissions. MRF and MBM carbon content is biogenic. Therefore, CO_2 emissions arising from their combustion do not contribute to the net gain of GHG in the atmosphere.

In the current study, CO_2 emissions associated with category 3 rendered products (i.e., MRF and PAP) were allocated using both mass and economic allocation, and system expansion to illustrate the effect of different co-product handling approaches. As the main purpose of rendering is to dispose of ABP with a charge to the producer; mass allocation is probably the most appropriate option. If economic allocation is used, the results would be dependent on variations in the price ratio between MRF and PAP (Figure 26). With system expansion it is not possible to obtain results for both category 3 MRF and PAP simultaneously.

The average percentage of total thermal energy derived from the combustion of MRF by the rendering plants that participated in the study was 76%. Consequently, a scenario where 25% of thermal energy is derived from natural gas and 75% from MRF can be considered representative of the UK. In this case, the fossil CO_2 emissions associated with category 1 MRF at the rendering plant gate were −0.77 kg CO_2e/kg.

Using mass allocation the fossil CO_2 emissions associated with category 3 MRF and PAP were 0.15 and 0.15 kg CO_2e/kg respectively. Using economic allocation the CO_2 emissions associated with category 3 MRF and PAP are 0.28 and 0.06 kg CO_2e/kg respectively. The robustness of the results in relation to some important choices in the modelled system is presented in the Sensitivity Analysis.

There is a considerable difference in CO_2 emissions between category 1 and 3 MRF. For category 1 MRF negative CO_2e emissions are realized by the replacement of grid electricity with electricity produced from the combustion of MBM in FBC power plants. It should be noted that when more category 1 MRF is used as a fuel within the system, more MBM is coproduced and thus the system gains more credits from the avoidance of British electricity. In contrast, in the category 3 system the protein meal (PAP) is a valuable co-product and the effect of electricity production from combustion of MBM is not as important.

The fossil CO_2 emissions calculated for the life cycle inventory of tallow in the Ecoinvent database are higher than those in the current study, with combustion of natural gas being the main contributor. The database does not differentiate between ABP categories, but states that ABP are treated as wastes, and that the system does not include animal production or slaughtering. Consequently, the result calculated with the inventory in the Ecoinvent database is comparable with the system modelled, assuming 100% natural gas. For category 3 MRF, using this scenario and mass allocation CO_2 emissions would be 0.29 kg CO_2e/kg (Figure 28). This is considerably lower than that calculated using the Ecoinvent database. This difference can be explained by the fact that in the inventory in the Ecoinvent database the amount of thermal energy required to produce tallow is considerable higher than that reported in the current study.

Palm oil and MRF both consist of long chain fatty acids (C_{16-18}), which are used by the oleochemical industry (e.g.,soap manufacture) (Postlethwaite, 1995). The GHG emissions for palm oil have been reported by Reijnders and Huijberts (2008), who included CO_2 emissions from combustion of fossil fuel for agriculture, processing and logistics; loss of biogenic carbon through land use change; and CH_4 production from anaerobic digestion of palm oil waste. Allocation of CO_2 emissions was based on economics. The results ranged from 2.8 to 19.7 kg CO_2e/kg palm oil mainly depending on plantation practices. Schmidt (2010) reported results between 2.16 and 2.60 kg CO_2e/kg palm oil using different modelling and allocation approaches. The Ecoinvent database life cycle inventory also provides a figure for GHG emissions from palm oil in Malaysia. All of these results are higher than the CO_2 emissions for category 1 and 3 MRF obtained in the current study.

PAP has a similar protein content to soya bean meal (Sellier, 2003) and has the potential to replace soya bean meal in animal diets. Dalgaard *et al.* (2008) reported a consequential LCA of soya bean meal for inclusion in livestock production. Since soya oil is a coproduct of soya bean meal, system expansion was used to include the avoidance of palm and rape oil production. The system included agriculture and milling of soya bean in Argentina, marginal production of palm oil in Malaysia and rapeseed and barley in Denmark. It also included transport of soya bean meal from Argentina to The Netherlands. Greenhouse gas emissions were 0.72 kg CO_2e/kg soya bean meal. Similarly, the Ecoinvent database also provides inventories for soya bean meal from Brazil and the US, respectively. All of these results are higher than the CO_2e emissions for PAP obtained in the current study.

The main reason that mammalian rendered products in the UK have lower CO_2 emissions compared to substitute vegetable oils and protein meals is because they are treated as wastes within LCA methodology. Consequently, they do not carry any of the environmental burden associated with their production. In addition, a significant percentage of the thermal energy required for rendering is derived from combustion of MRF that does not contribute to fossil CO_2 emissions. The system also gains credits from the production of biogenic electricity from the combustion of MBM.

The UK rendering industry currently produces approximately 74 000 tonnes of category 3 MRF and 97 500 tonnes of mammalian PAP per annum (Personal communication with Stephen Woodgate, FABRA). Assuming that category 3 MRF and PAP are used as direct replacements for palm oil and soya bean meal in current applications, and using fossil CO_2e emissions associated with palm oil and soya bean meal derived from the Ecoinvent databases, the use of UK rendered products can be estimated to reduce CO_2 emissions by approximately 70 000 tonnes per annum (excluding emissions associated with land transformations). If higher CO_2 emission values for palm oil and soya bean meal are used then CO_2 emission reductions calculated would be significantly higher. As category 1 MRF is used as a fuel in the rendering industry, it is included in the production of category 3 rendered products. However, sufficient category 1 MRF is produced by the UK rendering industry to satisfy its energy requirements and still replace fats used in the production of biodiesel or other carbon intensive form of electricity generation. The estimated CO_2 emission avoidance associated with use of UK rendered products should be treated with caution as the palm oil and soya bean meal replaced by UK rendered products could be used elsewhere.

Chapter 5

The relative importance of slaughtering and animal by-product management in the life cycle greenhouse gas emissions of beef production in the UK

A manuscript based on the content of this chapter has been submitted to *Environmental Science & Technology* as:

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5 The relative importance of slaughtering and animal by-product management in the life cycle greenhouse gas emissions of beef production in the UK

5.1 Abstract

The objective of the study was to analyse the influence of slaughtering and animal by-products (ABP) management on the greenhouse gas (GHG) emissions of beef production. Two system boundary scenarios were modelled. The first system only included on-farm beef production, whilst the second system included on-farm beef production, slaughtering, ABP rendering, and the use of rendered products to replace alternative products. Slaughterhouse co-products handling was performed using economic and mass allocation. Two contrasting slaughtering yield scenarios were used and the system was tested with 100% of the energy use for ABP rendering being derived from either natural gas or tallow. When tallow is not used as a fuel for the rendering process it is used to produce biodiesel.

Inclusion of slaughtering and ABP management had a minimal impact on the GHG emissions of beef production. Between 1.1 and 1.4% of GHG emissions associated with beef production originated from the slaughtering process. Marginal negative GHG emissions were derived from ABP management, because rendered products avoid the production of products that are associated with higher GHG emissions. Animal production was responsible for more than 99.6% of the GHG emissions of beef meat.

5.2 Introduction

As a result of increasing pressure for environmental sustainability in food production, the environmental impact of beef (and other meat systems) production has frequently been studied using life cycle assessment (LCA) or similar assessment tools (e.g. *Carbon Footprint*) (Ogino *et al.*, 2004; Casey and Holden, 2006a; Williams *et al.*, 2006; Ogino *et al.*, 2007b; Vergé *et al.*, 2008; Edward-Jones *et al.*, 2009; Beauchemin *et al.*, 2010; Nguyen *et al.*, 2010a; Pelletier *et al.*, 2010b; Peters *et al.*, 2010; Cederberg *et al.*, 2011). It is estimated that the animal production sector is responsible for 18% of global greenhouse gas (GHG) emissions (Steinfeld *et al.*, 2006). It is therefore important to gain a holistic understanding of the GHG emissions associated with each stage of animal production to enable targeted GHG mitigation strategies to be implemented. In a recent review of LCA studies of beef, pork and chicken production in Organisation for Economic Co-operation and Development (OECD) countries, de Vries and de Boer (2010) compared the results of 16 studies where a cradle-to-farm-gate approach had been taken. The GHG emissions associated with beef, pork and chicken production ranged from 14.0 to 32.0, 3.9 to 10.0 and 3.7 to 6.9 kg CO_2e/kg meat (carcass weight), respectively. The GHG emissions associated with pork and chicken production were lower than for beef production because ruminants consume more energy per kg of meat than non-ruminants and produce methane (CH_4) as a result of enteric fermentation. Additionally ruminants tend to have a lower reproductive rate and longer generation interval.

Following animal production, the conversion of animals to meat results in the production of animal by-products (ABP). Animal by-products include hides, skins, hairs, feathers, hoofs, horns, feet, heads, bones, toe nails, blood, organs, glands, intestines, muscle and fat tissues, shells and whole carcasses (Meeker and Hamilton, 2006). Animal by-products are classified by European legislation into three categories (EC, 2009). A description of these categories and the treatment options available is provided in section 2.1.4. The most common treatment option for ABP in the UK is rendering. Between 2006 and 2008 animal production in the UK was approximately 3.4 million tonnes live weight per year (FAO, 2010). Over the same period, the UK rendering industry processed approximately 2 million tonnes of ABP per year (pers. comm. SL Woodgate, FABRA). The slaughter of animals and disposal of ABP by rendering can be considered an essential part of the meat production system. Consequently, it is important to understand the relative contribution of post farm-gate processes to the life cycle GHG emissions associated with meat production.

LCA studies of beef production normally adopt one of two approaches. Either the functional unit is expressed as 1) the mass of live weight leaving the farm gate (Ogino *et al.*, 2004; Casey and Holden, 2006a; Edward-Jones *et al.*, 2009; Pelletier *et al.*, 2010b), or

2) as the mass of saleable meat calculated from the killing out percentage (KoP) (Williams *et al.*, 2006). In both approaches only the environmental burden associated with animal production up to the farm gate is included. Post farm gate processes such as slaughtering, meat processing and disposal of ABP by rendering are not included. Consequently, a true insight into the GHG emissions associated with meat production is not provided. Peters *et al.* (2010) investigated the GHG emissions associated with beef production in Australia and included emissions associated with meat processing. However, it is unclear whether or how the treatment of ABP was included in their modelling.

The primary objective of the current study was to investigate the relative contribution of post-farm gate processes (i.e. slaughtering and ABP management) to the GHG emissions associated with beef production at the slaughtering plant gate. In Chapter 4 it was demonstrated that GHG emissions associated with the rendering of ABP to produce rendered products could be minimised by generating process heat from the combustion of category 1 mammalian rendered fat (MRF) as opposed to fossil fuels. Secondary objectives of this study were (i) to verify the influence of substituting MRF for natural gas during the rendering process and (ii) to analyse the effect of modelling two contrasting slaughtering yields, on the GHG emissions of beef production.

5.3 Materials and methods

5.3.1 System description and boundaries

The functional unit was 1 kg of beef meat at the slaughterhouse gate. Two system boundaries were modelled B1 and B2 (Figure 30). B1 included animal production up to the farm gate. B2 included (i) animal production, (ii) slaughterhouse processes, (iii) ABP rendering, (iv) biodiesel production from category 1 MRF and the avoidance of fossil diesel combustion, production and extraction, (v) electricity production from the combustion of meat and bone meal (MBM) in Fluidized Bed Combustion (FBC) power plants and the consequent avoidance of British grid electricity, (vi) the avoidance of palm oil production due to substitution with category 3 MRF, and (vii) the avoidance of soya bean meal production due to substitution with mammalian processed animal protein (PAP). Palm oil (a marginal source of fat) and MRF (beef tallow in particular) both consist of long chain fatty acids (C_{16-18}), which are used in the same applications by the oleochemical industry (e.g. soap manufacture) (Postlethwaite, 1995). Soya bean meal (a marginal source of protein) has a similar protein content to PAP (Sellier, 2003) and is therefore an obvious substitute in pet foods and animal diets. After leaving the farm it was assumed that live animals were transported by lorry to the slaughterhouse/meat processing plant. Oceanic transport of avoided palm oil and soybean meal to the UK was included in the category 3 ABP managing system.

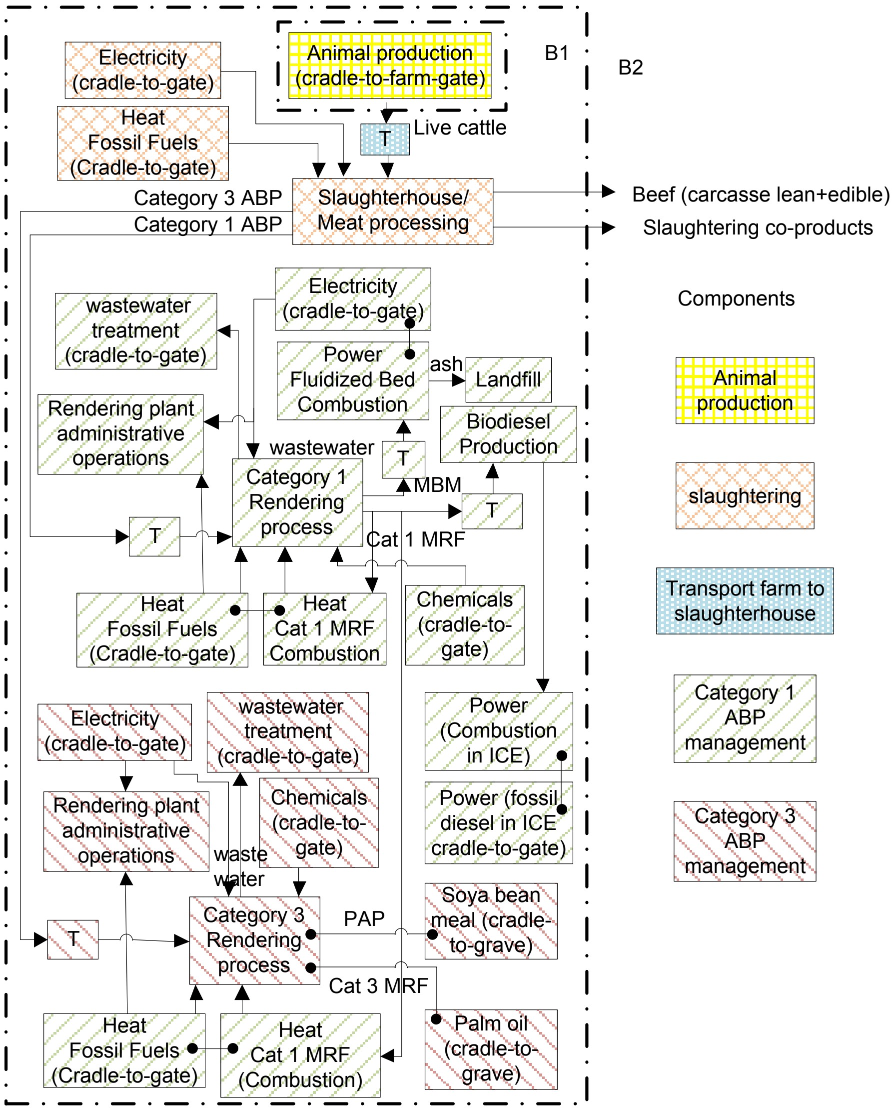

Figure 30 System Boundaries B1 and B2 for the beef production system (T: transport, ABP: animal by-product, MBM: meat and bone meal, MRF: mammalian rendered fat, PAP: processed animal protein, ICE: internal combustion engine)

5.3.2 Beef slaughtering yield cases

Two slaughtering yield scenarios reflecting industry practice were modelled, the best case and the worst case as defined by EBLEX (2006). In the best case the yield of edible material for human consumption was maximised. In the worst case, separation of edible and non-edible material was less efficient, such that a higher proportion of ABP was produced. The relative proportion of different co-products and by-products produced from each scenario are presented in Table 15.

Table 15 Relative percentage of each co-products and by-product produced during the slaughtering of beef cattle using the best and worst case scenario as defined by EBLEX (2006)

Slaughtering stream	Best case	Worst case
carcass lean	33.1	32.1
edible material[a]	21.1	11.0
hide and skin	7.3	7.1
petfood	0.8	0.0
gut content	15.4	0.0
category 3ABP[b]	11.1	20.7
category 1ABP[b]	11.2	29.2
Total beef cattle mass	100.0	100.0

[a]edible material other than carcass lean
[b]ABP: animal by-products

5.3.3 Co-product handling

The environmental burden associated with beef production was allocated between beef (i.e. carcass lean and edible material) and slaughterhouse co-products using two approaches, namely mass and economic allocation. Different methods of co-product handling in LCA produce different results (Cederberg and Stadig, 2003; Flysjö *et al.*, 2011a). Mass allocation factors (Table 16) were calculated using data presented in Table 15.

Economic allocation factors (Table 16) were calculated using the average price of beef and hides and skins published in the Meat Trades Journal (Meat Trades Journal) between December 2006 and July 2009. During this period the price of beef was between 10 and 40 times higher than that of hides and skins (Figure 31). In September 2008 hide prices dropped drastically due to the global financial crisis demonstrating economic factors are variable over time, whilst mass factors remain constant. The prices of gut contents and pet food were assumed as 30 and 100 pounds sterling per tonne respectively (this estimation reflect prices of fish and poultry by-products).

Table 16 Mass and economic allocation factors (%) for each slaughterhouse co-product[a]

Co-products	Best yield case		Worst yield case	
	Mass allocation	Economic allocation	Mass allocation	Economic allocation
Beef/edible	69.724	95.420	85.874	94.922
Skins and hides	9.400	4.184	14.126	5.078
Pet food material	1.086	0.061	na	na
Gut content	19.790	0.335	na	na

[a] na: not applicable

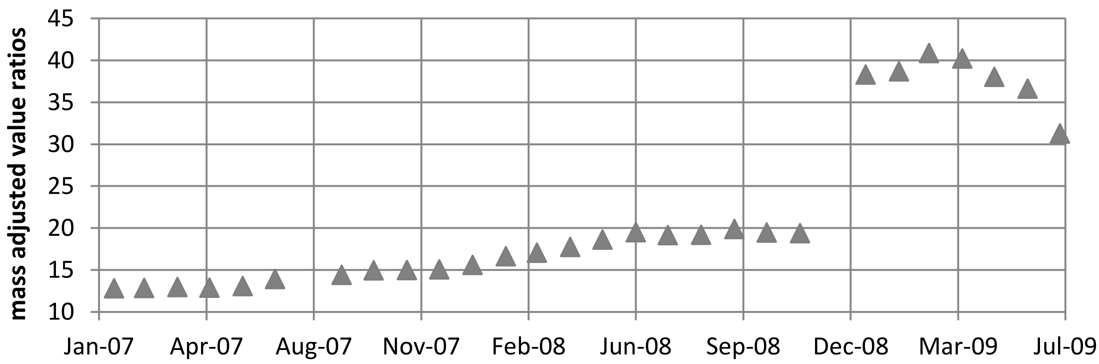

Figure 31 Mass adjusted price ratios of beef to hides and skins (Meat Trades Journal, 2007-2009)

5.3.4 Rendering fuel scenarios

The rendering industry uses both grid natural gas and category 1 MRF as fuels to produce process heat. If MRF is used as a fuel, there is a closed-loop recycling situation. The system was tested with 100% of the energy input being derived from either natural gas or MRF. The proportions of category 1 and 3 ABP produced varied depending on the slaughtering yields case (Table 15). In the case of 100% natural gas as fuel for rendering the entire category 1 MRF produced is used in the production of biodiesel. In the case of 100% MRF as fuel for rendering, part of the produced category 1 MRF is used as a fuel and the remainder is used in the production of biodiesel. Table 17 presents the distribution of final use of category 1 MRF for both slaughtering yield cases when MRF is used as fuel for rendering. Yields and energy requirements in rendering are based on data in Chapter 4.

Table 17 Amounts (kg) of category 1 mammalian rendered fat (MRF) used as a fuel in rendering and the production of biodiesel based on 1 kg of animal by-products (ABP) produced for both slaughtering yields cases when thermal energy for the rendering of ABP is provided entirely by category 1 MRF

	Best yield case	Worst yield case
cat 1 ABP produced[a]	1	1
cat 3 ABP produced	0.708	0.986
Final uses		
category 1 MRF used as fuel in rendering[b]	0.098	0.109
category 1 MRF used in the production of biodiesel	0.036	0.025

[a] The basis for presenting data is 1 kg ABP produced for both slaughtering cases, however the amounts of category 1 and 3 ABP produced in both slaughtering cases is different (Table 15).
[b] Energy intensity and yields in rendering are taken from Chapter 4

5.3.5 Data sources and calculation

GHG emissions associated with the production of beef production in the UK have been taken from Williams *et al*. (2006). The model was developed further under the DEFRA-funded project IS0222 and modified for this study to express the GHG emissions on a live weight rather than deadweight basis (pers. comm. AG Williams, 2011) as 6.888 kgCO_2e per kg live weight beef at the farm gate.

Data on energy use associated with slaughtering was derived from a study for Finland (The Finnish Environment, 2002). This reference provides heat and electricity use of 720 kJ and 684 kJ per kg carcass weight, based on average carcass weight of 260 kg. It is recognised that beef slaughtering and meat processing technology may vary from country to country. However, animal slaughtering in both Finland and the UK are regulated by EU legislation consequently and the process in both countries is similar.

Inventory data on category 1 and 3 rendering and its associated processes were taken from Chapter 4. The Ecoinvent database (Ecoinvent Centre, 2010) was used to provide inventories associated with the production of (i) British grid electricity, (ii) power from internal combustion engines, (iii) fossil diesel, (iv) soya bean meal from Brazil, (v) palm oil from Malaysia, and (vi) transport. Energy use in the production of biodiesel from MRF (beef tallow) was taken from Lopez *et al.* (2010).

Simapro 7.3 (PRe Consultants, 2011) was used for system modelling and calculation. Climate Change was assessed using the Greenhouse Gas Protocol 1.00 (The Greenhouse Gas Protocol, 2010) method which uses the climate change characterisation factors with a timeframe of 100 years reported by IPCC (Solomon *et al.*, 2007b). Carbon dioxide emissions associated with land transformation from soya bean meal in Brazil and palm oil production in Malaysia were included. Biogenic CO_2 emissions from the combustion of ABP were not included as it was considered that these do not represent a net gain of CO_2 in the atmosphere. For further information on the biogenic CO_2 emissions from the rendered products system see Chapter 4.

5.4 Results

The GHG emissions associated with 1 kg of beef for both system boundaries (B1 and B2) and for the best and worst slaughtering scenarios are presented in Table 18. The effects of allocation method and thermal fuel used during the rendering of ABP are also presented. The breakdown of the relative contributions of animal production, slaughtering and ABP management (Figure 30) to the total GHG emissions associated with beef production is also provided. When the best case slaughtering scenario was applied to system B1, the GHG emissions associated with the beef production were 8.882 and 12.156 kg CO_2e/kg for mass and economic allocation respectively. Conversely when the worst case scenario was applied, the GHG emissions increased to 13.763 and 15.214 kg CO_2e/kg beef respectively. When the best case slaughtering scenario was applied to system B2 and natural gas was used as fuel for the rendering of ABP, the GHG emissions associated with the beef production were 8.904 and 12.186 kg CO_2e/kg for mass and economic allocation respectively. Conversely when the worst case scenario was used GHG emissions increased to 13.549 and 14.976 kg CO_2e/kg beef respectively. When the fuel used for the rendering of ABP was category 1 MRF, the GHG emissions associated with beef production were marginally higher.

Table 18 Life cycle greenhouse gas emissions associated with 1 kg beef at the slaughtering gate[a]

Allocation	Fuel[b]	Units	Animal production	Slaughter	T[c]	ABPM[d]		Total
						Cat 1	Cat 3	
			Best yield scenario					
System boundaries B1								
mass	na	kg CO_2e	8.882	na	na	na	na	8.882
		%	100.0	na	na	na	na	100.0
economic	na	kg CO_2e	12.156	na	na	na	na	12.156
		%	100.0	na	na	na	na	100.0
System boundaries B2								
mass	natural gas	kg CO_2e	8.882	0.121	0.021	-0.043	-0.077	8.904
		%	99.8	1.4	0.2	-0.5	-0.9	100.0
mass	Cat 1 MRF[e]	kg CO_2e	8.882	0.121	0.021	-0.032	-0.077	8.915
		%	99.6	1.4	0.2	-0.4	-0.9	100.0
economic	natural gas	kg CO_2e	12.156	0.166	0.029	-0.059	-0.106	12.186
		%	99.8	1.4	0.2	-0.5	-0.9	100.0
economic	Cat 1 MRF[e]	kg CO_2e	12.156	0.166	0.029	-0.044	-0.106	12.201
		%	99.6	1.4	0.2	-0.4	-0.9	100.0
			Worst yield scenario					
System boundaries B1								
mass	na	kg CO_2e	13.763	na	na	na	na	13.763
		%	100.0	na	na	na	na	100.0
economic	na	kg CO_2e	15.214	na	na	na	na	15.214
		%	100.0	na	na	na	na	100.0
System boundaries B2								
mass	natural gas	kg CO_2e	13.763	0.149	0.032	-0.173	-0.224	13.549
		%	101.6	1.1	0.2	-1.3	-1.7	100.0
mass	Cat 1 MRF[e]	kg CO_2e	13.763	0.149	0.032	-0.132	-0.224	13.589
		%	101.3	1.1	0.2	-1.0	-1.6	100.0
economic	natural gas	kg CO_2e	15.214	0.165	0.036	-0.191	-0.248	14.976
		%	101.6	1.1	0.2	-1.3	-1.7	100.0
economic	Cat 1 MRF[e]	kg CO_2e	15.214	0.165	0.036	-0.146	-0.248	15.021
		%	101.3	1.1	0.2	-1.0	-1.6	100.0

[a] na: not applicable
[b] Fuel: Fuel used in the rendering process
[c] T: transport farm to slaughter
[d] ABPM: animal by-product management
[e] Cat 1 MRF: category 1 mammalian rendered fat

5.5 Sensitivity analysis

A sensitivity analysis was performed to test the choice of avoided products by the category 1 rendered products. In the base case system modelled, production of British average electricity is avoided by the combustion of MBM in FBC power plant, and surplus category 1 MRF is used in the production of biodiesel and avoids the production of fossil diesel and CO_2 emissions from its combustion in internal combustion engines (Figure 30). A test was performed assuming that the electricity avoided was produced from natural gas (marginal electricity) and that surplus category 1 MRF avoids the production of palm oil from Malaysia (similar to category 3 MRF as it could be argued that rendered fats displace not fossil fuels but other sources of marginal fat for the production of biodiesel). It should be noted that category 1 MRF can only replace palm oil in combustion applications. The sensitivity analysis was performed for system boundaries B2 (the only affected by changes in ABP management) and only for economic allocation at the slaughterhouse gate. Table 19 presents the results of this test.

Table 19 Life cycle greenhouse gas emissions associated with 1 kg beef at the slaughtering gate for avoidance of marginal products by both categories of rendered products[a]

Fuel[b]	Units	Animal production	Slaughter	T[c]	ABPM[d]		Total
					Cat 1	Cat 3	
Best yield scenario							
natural gas	kg CO_2e	12.156	0.166	0.029	-0.010	-0.106	12.235
	%	99.4	1.4	0.2	-0.1	-0.9	100
Cat 1 MRF[e]	kg CO_2e	12.156	0.166	0.029	-0.025	-0.106	12.220
	%	99.5	1.4	0.2	-0.2	-0.9	100
Worst yield scenario							
natural gas	kg CO_2e	15.214	0.163	0.036	-0.032	-0.248	15.134
	%	100.5	1.1	0.2	-0.2	-1.6	100
Cat 1 MRF[e]	kg CO_2e	15.214	0.165	0.036	-0.075	-0.248	15.092
	%	100.8	1.1	0.2	-0.5	-1.6	100

[a] na: not applicable
[b] Fuel: Fuel used in the rendering process
[c] T: transport farm to slaughter
[d] ABPM: animal by-product management
[e] Cat 1 MRF: category 1 mammalian rendered fat

This test illustrated that for either average or marginal products avoided, the contribution from the ABP management system to the GHG emissions of beef is negative and significantly minor. However the magnitude is different depending on the final use of the rendered products. Contrary to the base case, when the fuel used for the rendering of ABP was category 1 MRF, the GHG emissions associated with beef production were marginally lower.

5.6 Discussion and conclusion

The GHG emissions associated with beef production at the slaughtering plant gate depend mainly on the slaughtering scenario and co-product handling approach adopted. The inclusion of the slaughtering process and ABP management did not have a major effect on the GHG emissions of beef production. The choice of co-product handling approach is an important reason for differences in GHG emissions for similar product systems (Cederberg and Stadig, 2003; Ayer *et al.*, 2007; Curran, 2007a; Thomassen *et al.*, 2008a; Flysjö *et al.*, 2011a). When economic allocation is used the co-product with the highest economic value (beef) is allocated the greatest proportion of the GHG emissions associated with the system. When mass allocation is used GHG emission are allocated equally on a mass basis to all of the co-products. Consequently, GHG emissions of beef production at the slaughtering plant gate were lower when mass rather than economic allocation was used.

Within system boundary B2, on-farm animal production was responsible for greater than 99.6% of the GHG emissions of beef production (Table 18). The second largest positive contributor being slaughtering, which was responsible for 1.4 and 1.1% of the GHG emissions for the best and worst case slaughtering scenarios respectively. In both cases transport (of live cattle to the slaughterhouse) was responsible for 0.2% of GHG emissions.

The contribution of category 1 ABP management to the GHG emissions of beef production depended on the slaughtering scenario and fuel source used in the rendering process. The contribution of category 1 ABP rendering to the GHG emissions of beef were marginal and negative (between -0.4 and -1.3%). In the base case, the GHG emission credits were higher when natural gas was used as a fuel as opposed to MRF. This is because in the base system modelled, when category 1 MRF is not used as a fuel for rendering, it is used for the production of biodiesel and the conversion of more MRF to biodiesel to avoid fossil diesel resulted in a greater GHG avoidance than the use of MRF instead of natural gas as a fuel for rendering. In Chapter 4, it was reported the GHG emissions associated with mammalian rendered products and concluded that from a Climate Change perspective it was beneficial to maximise the use of MRF as a fuel for the rendering process. However, in this study the system boundaries were constrained to the ABP collection and the rendering process. The current study includes beef production at the slaughtering plant gate with the system being expanded to include total destruction of unavoidable ABP (rendering and use of rendered products). The contribution of category 3 ABP management to the GHG emissions of beef production was also minor and negative (between -0.9 and -1.7%), regardless of the fuel used in the rendering process. The negative emissions of category 3 ABP rendering are due to the avoidance of marginal palm oil and soybean meal production (which incur emissions associated to agricultural processes and land transformation).

It is important to notice the effect of replacing not fossil diesel (and associated production and fossil emissions from combustion), but replacing a different source of marginal fat to produce biodiesel (as in the sensitivity analysis). In this test, the system does not gain credits from fossil diesel combustion (and production) avoidance since both MRF and palm oil are biogenic material. The credits for category 1 MRF in the sensitivity test are associated with the avoidance of agricultural and land transformation emissions from palm oil (as with the category 3 MRF). Contrary to the best case, in the sensitivity analysis, beef production performed better from a Climate Change perspective when category 1 MRF was used as fuel in rendering (instead of natural gas). Most importantly, ABP management provided GHG emission credits in any case (Tables 18 and 19).

GHG emissions associated with the beef production at the slaughtering plant were lower for the best than the worst case slaughtering scenario. With higher yields, less animal live weight is required to produce a given amount of beef. However, the GHG emission avoidance from ABP management was higher for the worst case slaughtering scenario because more material that avoids the production of higher GHG intensity substitutes (fossil fuels, soya bean meal and palm oil) is produced. Higher yields reduce GHG emissions from on farm beef production, which is ultimately the most important component in the beef production system. But higher yields may result in more domestic waste production without necessarily an increase in food intake (e.g. more bone and/or fat). This indicates that an even more holistic and detailed approach including consumption in households and domestic waste management is required to adequately analyse the effect of slaughtering yields and animal by-product processing.

The primary energy requirements for slaughtering have been reported as 1390 kJ/kg carcass weight and 5500 kJ/kg final meat product (Ramírez *et al.*, 2006). Similarly, Lopez *et al.* (2010) reported that the energy requirement for slaughtering ranged from 1323 – 5291 kJ/kg (15% electricity + 85% heat) carcass weight. These values are in good agreement with the value of 1404 kJ/kg in The Finnish Environment (The Finnish Environment, 2002) used in the current study. If the value at the higher end of the range reported by Lopez *et al.* (2010) had been used in the system modelled, with economic allocation and natural gas as a fuel for rendering then GHG emissions would have increased to 12.738 and 15.710 kg CO_2e/kg beef for the best and worst case slaughtering scenarios respectively. In this situation the contribution from slaughtering would increase to 5.6 and 5.7% for best and worst case slaughtering scenarios respectively. The relative contribution of slaughtering to the GHG emissions of beef production is strongly dependent on energy use during slaughtering. However, animal production is still the greatest contributor.

Peters *et al.* (2010) suggested that approximately 10% of the GHG emissions associated with beef production in Australia were due to meat processing. This is considerably higher than in the current study. Differences in energy use during slaughtering may be the reason. In the study of Peters *et al.* (2010) electricity and thermal energy use was considerably higher than those used in the current study (1440 and at least 1300 kJ/kg carcass weight respectively considering an energy density of 24.5 MJ/kg coal (DEFRA, 2010b)). Peters *et al.* (2010) also included additional thermal energy from LPG that has not been accounted in the comparison here. In the current study all the heat in slaughtering is derived from natural gas, which has a lower GHG emission intensity than coal, which is one more reason for the lower Climate Change relevance from slaughtering.

The GHG emissions from on-farm beef production used in the current study were 6.888 kgCO_2e/kg beef live weight (pers. comm. AG Williams, 2011), which represents an average figure for England and Wales based on system modelling (Williams *et al.*, 2006). Empirical studies based on real farm data have reported GHG emissions of 15.5 and 13.0 kgCO_2e/kg beef live weight for conventional systems in Wales (Edward-Jones *et al.*, 2009) and Ireland (Casey and Holden, 2006a) respectively. The value used in the current study is relatively low and it should be noticed that if higher GHG intensity for beef production would have been used, the contribution of slaughtering and ABP management would have been even lower.

The current study has demonstrated that the inclusion of animal slaughtering and ABP rendering in the system boundaries does not radically affect the GHG emissions associated with beef production. However slaughtering yields and co-product allocation methods do. Maximising the utilisation of produced beef (higher yield) results in lower GHG emissions associated with the mass of edible product. The inclusion of slaughtering adds positive GHG emissions to beef production, but these are very low in comparison with on-farm GHG emissions. The treatment of ABP by rendering provides credits to the beef production system through the avoidance of products such as fossil fuels, and marginal protein meals and fats. The benefits of ABP by rendering are marginal in comparison to the emissions from the animal production system.

Chapter 6

Life cycle fossil energy use and greenhouse gas emissions of broiler production in the UK

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6 Life cycle fossil energy use and greenhouse gas emissions of broiler production in the UK

6.1 Abstract

Chicken meat derived from broiler production accounted for approximately 38 % of total UK meat production between 2000 and 2009. To date, there have been limited studies that comprehensively investigate the environmental burden of broiler production. The objectives of the study were to quantify the fossil energy use and greenhouse gas (GHG) emissions associated with two broiler production systems in the UK producing birds of different weights ('Standard' and 'Heavy') and to identify mitigation strategies. The study employed generic life cycle assessment methodology, using industry data to model economic flows and databases to account for natural flows.

For the 'Standard' production system fossil energy use and GHG emissions were 8,961 MJ and 1,798 kg CO_2e/tonne live weight at the farm gate. For the 'Heavy' production system fossil energy use and GHG emissions were: 9,594 MJ and 1,901 kgCO_2e/tonne live weight. In both production systems, feed production and delivery was the greatest contributor to fossil energy use and GHG emissions, contributing 62 % and 79 % respectively. GHG emissions associated with UK broiler production are estimated to be 3.5 million tonnes CO_2e (approximately 0.6 % of total UK GHG emissions). Greatest mitigation opportunities for reducing the GHG emissions associated with broiler production are related to feed production (reduction in feed conversion ratio (FCR), changes in feed formulation, mitigation in crop agriculture) and improving energy efficiency in broiler houses.

6.2 Keywords

Broiler chicken, feed, life cycle assessment, energy, greenhouse gases, mitigation

6.3 Introduction

It is widely accepted that agriculture has a significant impact on the environment. Steinfeld *et al.* (2006) estimated that animal production accounts for 18 % of global greenhouse gas (GHG) emissions, whilst within Europe, livestock farming is estimated to be responsible for 10 % of GHG emissions (Lesschen *et al.*, 2011). The majority of GHG emissions from EU livestock farming originate from beef, milk and pork production which are estimated to be responsible for 29, 29 and 25 % of livestock GHG emissions, respectively (Leip *et al.*, 2010). Within Europe, poultry meat production is estimated to be the fourth largest contributor (8 %) to GHG emissions arising from livestock production.

The term poultry includes chickens, turkeys, geese and ducks. In the UK, broiler chickens are the predominant poultry meat produced, with an average of 835 million birds slaughtered per annum between 2000 and 2009 (FAO, 2011), representing approximately 38 % of total meat production. Globally, meat production is projected to more than double from 1999-2001 to 2050, with poultry being the commodity of choice (Steinfeld *et al.*, 2006). Indeed, between 2000 and 2008 global meat production increased from 233 to 279 million tonnes, an increase of 19%. Over the same period, the global chicken stock increased from 14.5 to 18.4 thousand million, an increase of 26 % (FAO, 2011). Consequently, it is important to quantify the GHG emissions associated with broiler production.

Life Cycle Assessment (LCA) is a mature ISO-standardised tool for evaluation of the environmental impact of a product or service throughout its life cycle (ISO, 2006a; b), which may include various impact indicators (climate change, eutrophication, acidification, etc). Recently, life cycle approaches dedicated only to climate change, often called *Carbon Footprints* (CF), have become frequent as standardised by the British Standard (BSI, 2008b). Life cycle approaches (LCA or CF) have been used to evaluate the environmental impact of ruminant products such as milk and beef (de Boer, 2003; Casey and Holden, 2005; 2006a; Williams *et al.*, 2006; Thomassen *et al.*, 2008b; Edward-Jones *et al.*, 2009; Beauchemin *et al.*, 2010; Pelletier *et al.*, 2010b; Peters *et al.*, 2010; Flysjö *et al.*, 2011b). Similarly, studies have been conducted on non-ruminant products such as pig meat (Basset-Mens and van der Werf, 2005; Williams *et al.*, 2006; Pelletier *et al.*, 2010a) and poultry meat (Williams *et al.*, 2006; Katajajuuri *et al.*, 2008; Pelletier, 2008). However, there are notably fewer studies relating to non-ruminant than ruminant products, possibly because ruminant products are associated with significantly higher GHG emissions because of enteric fermentation in the rumen (de Vries and de Boer, 2010). In spite of their lower GHG emission intensity, pig and poultry production are important sectors of both the UK and global animal production industry, and it is important to understand their contribution to global warming. Research into the environmental impacts of poultry production has normally focussed on point-source emissions relating to litter and gaseous emissions from farms, with holistic life cycle approaches being less frequent (Pelletier, 2008). Life cycle approaches are important because they help to identify elements in a production chain where the greatest improvements can be made.

GHG emissions associated with broiler production, derived from data collected from commercial farms have been reported for both the US and Finland (Katajajuuri *et al.*, 2008; Pelletier, 2008). Similarly, GHG emissions associated with broiler production, derived using a system modelling approach have been reported for the UK (Williams *et al.*, 2006). However, no studies have reported the GHG emissions for UK broiler production derived from data collected from commercial farms.

The primary objective of the current study was to collect data from commercial farms to quantify the life cycle fossil energy use and GHG emissions associated with two broiler production systems in the UK. The secondary objective was to examine the fossil energy use and GHG emissions associated with broiler feed production.

6.4 Materials and methods

The study employed a generic LCA methodological framework (ISO, 2006a; b). Two functional units were employed for each production system: (i) 1 tonne broiler live weight at the farm gate, and (ii) 1 tonne broiler feed delivered to the farm. Data collected from commercial farms and a feed mill was used to quantify economic flows (product and waste flows) in both the broiler production systems, and in the broiler feed system. Data for economic flows was supplied by VION Food Group Ltd (VION). Infrastructure processes (e.g. production of equipment, maintenance and repair of agricultural vehicles) were excluded from the product systems.

The co-product handling method employed depended on the situation and co-products analysed. Economic allocation was preferred in cases of co-production of feed ingredients. System expansion was used to account for the burdens avoided by utilising poultry litter, wash water, and rendering of mortalities.

The system was modelled using the Simapro 7.3 ® software package (PRe Consultants, 2011), which provides assistance in calculation and access to databases. Fossil energy use was assessed using the Cumulative Energy Demand 1.08 impact assessment method (Frischknecht *et al.*, 2003). Climate Change was assessed using the Greenhouse Gas Protocol 1.01 impact assessment method (The Greenhouse Gas Protocol, 2010). The product systems are described as follows:

6.4.1 Broiler production system

Two different broiler production systems were studied, which are representative of 95% of the indoor reared chickens produced according to independently audited standards in the UK (RSPCA, 2008). In the first system (Standard) both male and female birds were reared to 2.2 kg live weight, whilst in the second system (Heavy), female birds were reared to 2.2 kg and male birds were reared to 3.9 kg live weight**.** The characteristics of each production systems are presented in Table 20. Slightly different feed formulations were used in each production system (Table 21).

Economic flows were collected for two commercial farms for each production system during 2010. Altogether the 4 farms produced 8,306 tonnes live weight during the study period. The ‘Standard’ farms produced 3,872 and the ‘Heavy’ farms produced 4,434 tonnes live weight, which represents 0.47 % of annual broiler production in the UK (assuming 835 million head of chicken are slaughtered annually between 2000 and 2009 (FAO, 2011) at an average slaughter weight of 2.1 to 2.2 kg per head).

Table 20 Economic flows and summary information associated with 'Standard' and 'Heavy' broiler production systems in the UK (per 1 tonne live weight)

Economic flow or attribute in general information	Standard	Heavy
General information		
Average start weight (g)	43.00	43.00
Stocking density (Birds/m^2)	19.34	16.74
Average cycle length (days)	48.79	59.75
Average slaughter weight (kg)	2.24	3.06
Ventilation and lighting control system type	ON-OFF	ON-OFF
Inputs		
Feed Conversion Ratio (FCR) [a]	1.72	1.84
Chicks mass (kg)	19.89	14.63
Electricity (MJ)	335.05	420.48
Heating (MJ) [b]	1,820.73	1,805.47
Diesel (MJ) [c]	28.32	28.32
Bedding (shavings) (kg)	71.12	62.59
Cleaning chemicals (kg) [c]	0.75	0.75
Water use (kg)	3,751.87	3,573.22
Outputs		
Mortalities (kg)	8.55	12.90
Wash water (m^3)	0.15	0.21
Litter (kg)	580.8	610.8
Rubbish (kg) [c]	3.33	3.33
CH_4 from manure management (kg) [c,d]	0.88	0.88
N_2O from manure management (kg) [c,d]	0.36	0.36

[a] Feed Conversion Ratio (FCR), defined as kg feed consumed divided by kg live weight gained
[b] Conversion of volume of LPG using a density of 1,968 litres/tonne and a Net Calorific Value 45.91 GJ/tonne (DEFRA, 2010b)
[c] Global average (not dependent on production system)
[d] Natural flows

Table 21 Aggregated diet formulations used in the 'Standard' and 'Heavy' broiler production systems

Ingredient	Standard (%)	Heavy (%)
Wheat	49.56	51.02
Wheat (added non milled)	11.41	11.23
Soya meal	21.51	20.33
Field beans[a]	7.24	7.34
Rapeseed[a]	5.92	6.00
Soya oil	3.60	3.52
Limestone	0.50	0.37
Monocalcium phosphate	0.14	0.10
Fishmeal	0.11	0.09

[a]Field beans and rapeseed are milled together and deliver as a single vegetable protein meal to the feed mill

The system modelled is presented in Figure 32. The system includes*:* (i) chick production and delivery (ii) feed production and delivery to the farms (see section 2.2), (iii) heating of the broiler houses (liquefied petroleum gas (LPG)), (iv) transport of LPG to the farms (assumed 200 km), (v) weekly trial tests of emergency diesel generators, (vi) transport of diesel to the farms (assumed 200 km), (vii) production of British electricity, (viii) production of bedding material, (iix) transport of bedding material to the farm (assumed 200 km), (ix) water use, (x) production of chemicals, (xi) transport of chemicals to the farm (assumed 200 km), (xii) methane and nitrous oxide emissions resulting from manure management (xiii) transport of litter to secondary farms for land spreading (assumed 10 km), (xiv) transport of wash water to secondary farms for land spreading (assumed 10 km), (xv) litter disposal scenarios, (xvi) avoided inorganic fertiliser production due to displacement by wash water, (xvii) transport of mortalities to the rendering plant (assumed 200 km), (xviii) rendering of mortalities in category 1 or 2 rendering plant, (xix) transport of waste to landfill, and (xx) landfill. The final use of category 2 rendered products arising from the rendering of mortalities was included in the system (Figure 32) according to economic flows reported by Chapter 4 and Lopez *et al.* (2010). It was assumed that category 1 rendered fat produced was used as fuel in the rendering process and the surplus rendered fat in the production of biodiesel. Under EU legislation (EC, 2009) poultry mortalities are classified as category 2 animal by-products however they are normally mixed with, and processed as category 1 animal by-products. Transport distances of 200 km for inputs were considered to be representative of UK broiler production. GHG emissions as methane (CH_4) and nitrous oxide (N_2O), associated with manure management were calculated using a Tier 1 approach as presented in the IPCC guidelines (2006), and the emission values used are presented in Table 20. In the system modelled, the only flows considered in relation to chick production were energy use (gas oil and electricity). The amount of heat (0.14 MJ per chick) and electricity (0.24 MJ per chick) used in hatcheries are aggregated figures provided from 4 VION hatcheries in UK. Data sources for the development of the life cycle inventory of broiler production are presented in Table 22.

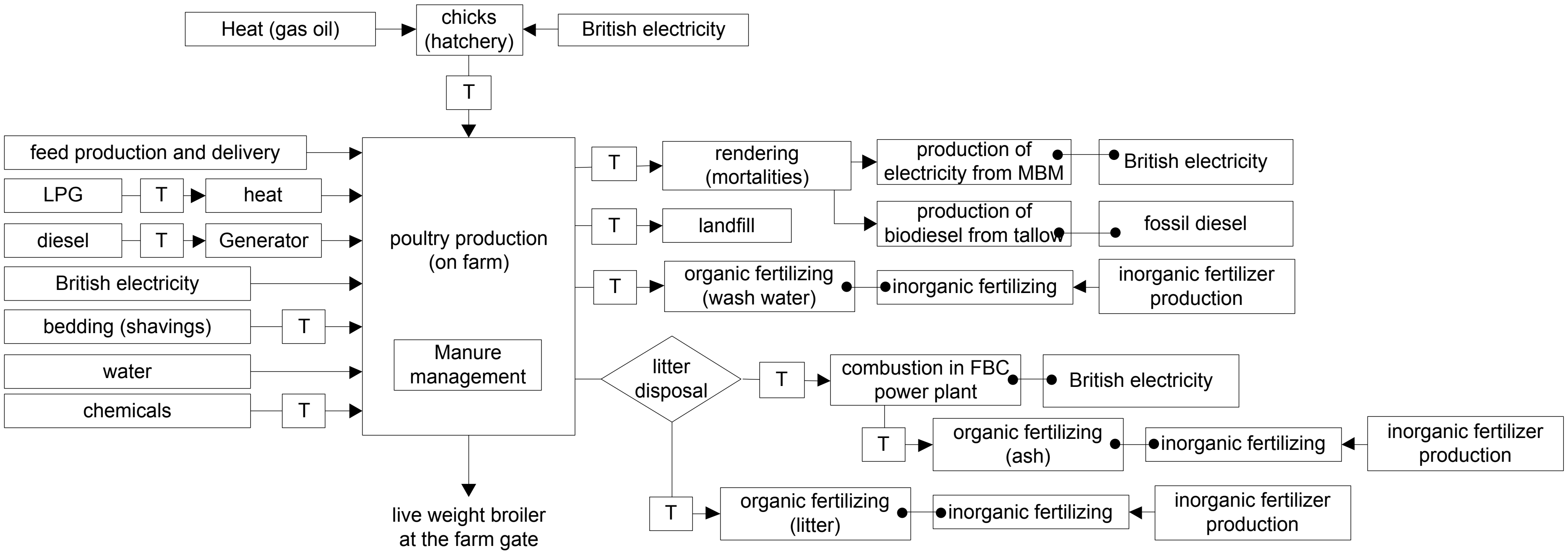


Figure 32 Broiler production system showing main processes (T: transport)

Table 22 Data sources used in the life cycle inventory for broiler production flows

Process	Data source	Geographical relevance
Heat from LPG	Production of LPG: Ecoinvent databases (Ecoinvent Centre, 2010) GHG emissions from LPG combustion: DEFRA (2010b)	Production: Europe GHG emissions: UK
Heat from natural gas	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland. Modified for UK natural gas production
Diesel generator	Ecoinvent databases (Ecoinvent Centre, 2010)	Europe
Electricity (average)	Ecoinvent databases (Ecoinvent Centre, 2010)	UK
Electricity from natural gas power plants (marginal)	Ecoinvent databases (Ecoinvent Centre, 2010)	UK
Bedding (production of shavings)	US Life Cycle Inventory (NREL, 2008)	US (the only life cycle inventory process available for this process, technology is probably similar)
Water	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland modified for UK electricity
Chemicals	Ecoinvent databases (Ecoinvent Centre, 2010)	Europe
Inorganic fertilisers production	Ecoinvent databases (Ecoinvent Centre, 2010)	Europe
rendering (and system expansion for heat for the rendering process, electricity, and biodiesel production from remaining tallow)	Economic flows adapted from Chapters 4 and 5 Biodiesel production adapted from Lopez *et al.* (2010)	UK (rendering) US (biodiesel production)
Landfill	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland (assumed similar to UK)
Emissions from manure management	Calculation according to Tier 1 IPCC (2006)	
Emissions from organic fertilising	Calculation according to Tier 1 IPCC (2006)	
Emissions from inorganic fertilising	Calculation according to Tier 1 IPCC (2006)	
FBC power plant electricity generation	Global efficiency (Yassin *et al.*, 2009)	UK
Transport	Ecoinvent databases (Ecoinvent Centre, 2010)	Europe

The nutrient value of wash water was determined by analysing two samples of wash water from one farm to determine the ammonium nitrogen, phosphorus (P_2O_5) and Potassium (K_2O) content (Table 23). Indirect GHG emissions associated with volatilization of N in organic fertiliser were included. System expansion was used to model the avoidance of inorganic fertiliser use including: (i) indirect GHG emissions associated with volatilization of N from inorganic fertiliser, and (ii) production of the inorganic fertiliser. Indirect emissions of N_2O associated with volatilization were included because these vary depending on whether the fertiliser applied is inorganic or organic. Emissions associated with the application of organic fertiliser to land were calculated as 0.002 kg N_2O per kg of N in litter and wash water. Emissions associated with the application of inorganic fertilisers to land were calculated as 0.001 kg N_2O per kg of N in fertiliser using a Tier 1 approach as detailed in the IPCC guidelines (2006). Poultry litter and wash water is normally spread on farms located near to broiler farms, hence a transport distance of 10 km was assumed.

Table 23 Nutrient content of wash water and poultry litter, nutrient availability factors and values used for the avoidance of inorganic fertiliser production

Parameter	Value from literature	Samples average	Availability factor	Value used as avoided production
Wash water				
Ammonium – N (kg/m^{3})	n.a.[a]	0.195	100%[d]	0.195
Phosphorus - P_2O_5 (kg/m^3)	n.a.[a]	0.464	100%[d]	0.464
Potassium - K_2O (kg/m^3)	n.a.[a]	0.504	100%[d]	0.504
Litter				
Dry matter (%)	35.8[b]	32.7	n.a.[a]	n.a..[a]
Readily available - N (kg/tonne fresh weight)	10[b]	4.6	100%[e]	4.5
Phosphorus - P_2O_5 (kg/tonne fresh weight)	25.2[b]	16.4	60%[f]	9.8
Potassium - K_2O (kg/tonne fresh weight)	19.3[b]	22.8	90%[f]	20.5
Hydrogen – H (% dry matter)	4.6[c]	5.8	n.a.[a]	n.a.[a]
Gross Heating Value (MJ/tonne dry matter)	13.1[c]	18.4	n.a..[a]	n.a.[a]
Ash content (% dry matter)	33.7	-	100%[g]	n.a.[a]

[a] n.a. Not applicable
[b] Nicholson *et al*. (1996)
[c] Quiroga *et al.* (2010)
[d] expert judgement (Paul Lewis, pers. comm., 2011)
[e] 100% since this N is readily available
[f] DEFRA (2010a)
[g] Fibrophos (2011)

6.4.2 Litter disposal scenarios

Two scenarios for litter disposal were modelled: Litter-to-fertiliser and Litter-to-power. Over 60% of the litter produced by VION was sent to power plants whilst the rest is sent to land spreading. Thus the main scenario is litter-to-power.

Litter-to-fertiliser: This scenario involved the disposal of litter by land spreading as organic fertiliser. Litter disposal was treated in the same way as wash water (see above). The nutrient value of litter was determined by taking two samples of litter from one farm and analysing to determine the ammonium nitrogen, phosphorus (P_2O_5) and Potassium (KO_2) content. The results were in agreement with literature values as presented in Table 23. Availability factors were used to account for inorganic replacement (Table 23).

Litter-to-power: This scenario involved the disposal of litter by combustion in Fluidised Bed Combustion (FBC) power plants for the production of electricity. Litter was transported to a FBC power plant (assuming a distance of 200 km), where the material was burnt to produce electricity. The production of electricity by this means displaces the production of average British electricity. A test with natural gas as the electricity technology displaced is presented in the Sensitivity Analysis.

Litter samples were also analysed for dry matter (DM), Gross Heating Value (GHV) and H content. The Net Heating Value (NHV) on a DM and fresh weight basis, were calculated according to Quiroga *et al.* (2010). The values are in reasonable agreement with literature values as presented in Table 23.

It was assumed that the conversion efficiency of FBC power plants is 18 % (Yassin *et al.*, 2009). Thus, in the life cycle inventory, the combustion one tonne of fresh poultry litter in a FBC power plant would produce 1,982 MJ of electricity.

The ash produced from FBC power plants is used as a fertiliser (Fibrophos). Using the ash content of broiler litter reported by Quiroga *et al.* (2010) and the DM content of poultry litter (Table 23), the ash content of fresh poultry litter was calculated as 12%. The Nitrogen, Phosphorus and Potassium contents of Fibrophos were 0, 12 and 12%

respectively (Fibrophos, 2011). Land spreading of Fibrophos avoids the production of inorganic fertilisers. Ash was transported to land spreading assuming a distance of 200 km.

6.4.3 Broiler feeds production and delivery system

Feed production starts with the production of ingredients. Once ingredients are delivered to the feed mill, there are two stages involved in feed production: 1) milling, mixing and pelleting (milling) and 2) blending in whole wheat (Figure 33). Broiler production involves five growth phases, with different diets being offered during each phase. Diet formulations for each phase were provided by VION and the composition of aggregate diets is presented in Table 21.

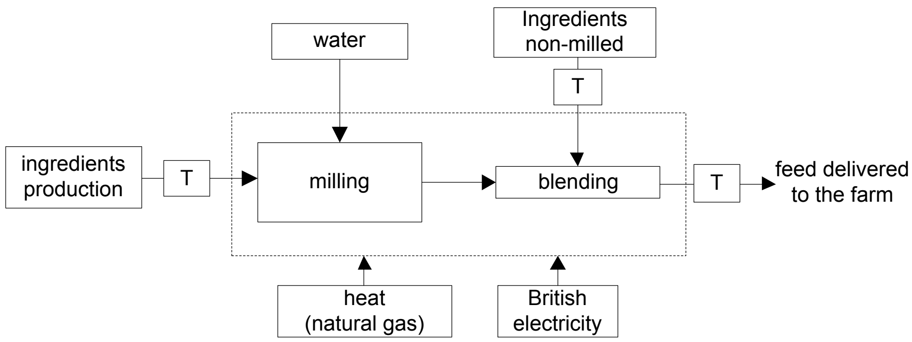


Figure 33 Broiler feed production and delivery system showing main processes (T: transport)

Data was supplied for a commercial broiler feed mill for 2010, which produced 275,300 tonnes/annum. The amount of energy used in feed production (MJ/tonne) was 147.79 MJ heat (as energy contained in the fuel: natural gas) and 65.3 MJ electricity (delivered to the plant) (Figure 33). The amount of water used in heat treatment (steam) during milling was 66.9 kg/tonne milled feed produced. Transport of ingredients to the feed mill, and feed from the plant to broiler farms, was assumed to be 200 km for each journey.

Field beans and rapeseed were milled together and delivered to the feed mill as a single vegetable protein meal containing 55 % field beans and 45 % rapeseed. Data on water and electricity used in the production of this vegetable protein meal was supplied by VION.

Data sources for the development of the live cycle inventory of broiler feed production and delivery are presented in Table 24.

Table 24 Data sources used in the life cycle inventory for broiler feed production and delivery flows

Process	Data source	Geographical relevance
Wheat (intensive production)	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland (assumed similar)
Fishmeal[a]	LCA foods database (Nielsen *et al.*, 2003)	Denmark (assumed similar)
Soya meal[b]	Production: Ecoinvent databases (Ecoinvent Centre, 2010) Processing: LCA foods database (Nielsen *et al.*, 2003)	Production of soya bean: Brazil Processing: Denmark (assumed similar)
Soya oil[c]	Production: Ecoinvent databases (Ecoinvent Centre, 2010) Processing: LCA foods database (Nielsen *et al.*, 2003)	Production of soya bean: Brazil Processing: Denmark
Limestone	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland (assumed similar)
Monocalciumphosphate	LCA foods database (Nielsen *et al.*, 2003)	Denmark (assumed similar)
Fava beans (intensive production)	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland (assumed similar)
Rapeseed (intensive production)	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland (assumed similar)
Water	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland modified for UK electricity
Heat from natural gas	Ecoinvent databases (Ecoinvent Centre, 2010)	Switzerland modified for UK natural gas production
Electricity (average)	Ecoinvent databases (Ecoinvent Centre, 2010)	UK
Transport	Ecoinvent databases (Ecoinvent Centre, 2010)	Europe

[a] Modified to allocate part of the environmental burden and resource use to the co-product fish oil according to the mass fraction of the co-products. Originally the life cycle inventory included system expansion that included the avoidance of rapeseed oil production. This is the only feed ingredient that allocation is based on mass as no economic information was available. The inclusion rate of this ingredient is not high and it is not expected radical variations in final results if using economic allocation.
[b] Modified to allocate part of the environmental burden and resource use to the co-product soya oil according to the economic fraction of the co-products. Economic information was taken for years 2006 to 2010 from The World Bank (2011). Originally the life cycle inventory included system expansion that included the avoidance rapeseed oil production.
[c] Modified from original life cycle inventory of Soya meal ([b]) that included system expansion for the avoidance of rapeseed oil production. System was modified to obtain soya oil also as a co-product of the system according to economic fraction of the co-products.

The Ecoinvent database (Ecoinvent Centre, 2010) inventory for soya bean production in Brazil includes the non-renewable biogenic emissions associated with land transformation, using the proportion of land use transformation in the previous 5 years from arable land, shrub land and tropical rainforest. In the source the emissions from clear cutting of primary forest are fully allocated to the provision of stubbed land for cultivation that can be used for 2 years.

6.5 Results

6.5.1 Life cycle energy use in broiler production

Fossil energy used in the two broiler production systems studied is presented in Figure 34. Using the litter-to-power scenario, the total amount of fossil energy used to produce one tonne broiler live weight in both the 'Standard' and 'Heavy' production systems was 8,961 and 9,594 MJ, respectively. Using the litter-to-fertiliser scenario the total amount of fossil energy used to produce one tonne broiler live weight in both the 'Standard' and 'Heavy' production systems was 11,127 and 11,871 MJ, respectively. Both litter disposal scenarios resulted in negative energy use and therefore provided credits to the broiler production system.

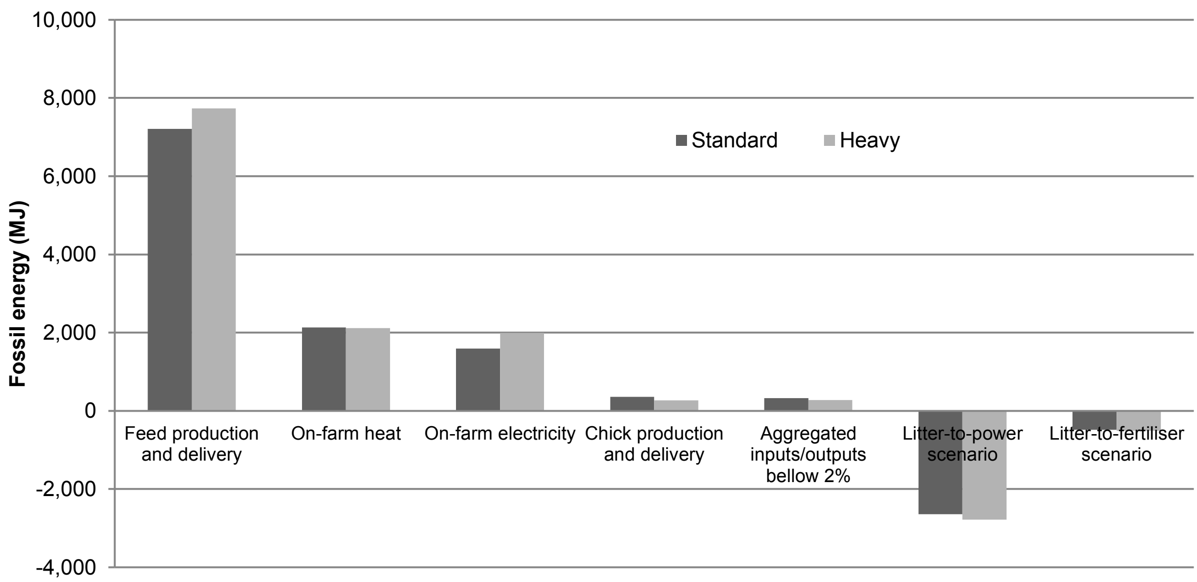


Figure 34 Fossil energy use associated with 'Standard' and 'Heavy' broiler production systems in the UK (MJ/tonne live weight)

The energy used in feed production and delivery accounted for the highest contribution to total energy consumption in the system. On-farm heat and electricity use were the second and third highest contributors respectively, whereas, chick production and delivery was the fourth highest energy user. These four accounted for 97 % of the fossil energy consumption in the broiler production systems studied when litter disposal was excluded (Figure 34).

6.5.2 Life cycle GHG emissions associated with broiler production

The GHG emissions associated with the two broiler production systems studied are presented in Table 25. Using the litter-to-power scenario the total GHG emissions associated with the production of one tonne broiler live weight in the 'Standard' and 'Heavy' production systems were 1,798 and 1,901 kg CO_2e, respectively. Using the litter-to-fertiliser scenario the total GHG emissions were 1,976 and 2,088 kg CO_2e, respectively.

The GHG emissions associated with feed production provided the highest contribution to total emissions for the broiler production systems studied (79 %). The second highest contributor was on-farm heat, with manure management and on-farm electricity being the third and fourth highest contributors respectively. These four accounted for approximately 98 % of the total GHG emissions associated with broiler production when litter disposal was excluded (Table 25).

Negative GHG emissions arise from the rendering of mortalities and their associated credits (Chapter 4) and from litter disposal. Both litter disposal scenarios resulted in negative emissions, with the greatest credits being gained from the litter-to-power scenario. Spreading wash water to land also avoids GHG emissions associated with inorganic fertiliser production. However, these reductions were lower than the GHG emissions associated with transport of the wash water to where it will be applied to land. Consequently, application of wash water to land is associated with marginally positive GHG emissions.

Table 25 Greenhouse gas emissions associated with 'Standard' and 'Heavy' broiler production systems in the UK (kg CO_2e/tonne live weight)

Input/output	Fossil ($kgCO_2e$)	Land transformation ($kgCO_2e$)	Total ($kgCO_2e$)	Contribution (%)
Standard				
Feed production and delivery	1,163	424	1,586	79.04
On-farm energy use (Heat and electricity)	242	0	242	12.08
Manure management (emissions)	129	0	129	6.44
Chick production and delivery	26	0	26	1.34
Others minor contributors	23	0	23	1.19
Rendering of mortalities (including transport)	-2	0	-2	-0.10
Total (excluding litter disposal)	1,584	424	2,007	100.00
Litter-to-power scenario	-209		-209	
Total (litter-to-power scenario)	1375		1,798	
Litter-to-fertiliser scenario	-31		-31	
Total (litter-to-fertiliser scenario)	1553		1976	
Heavy				
Feed production and delivery	1,249	434	1,683	79.41
On-farm energy use (Heat and electricity)	268	0	268	12.65
Manure management (emissions)	129	0	129	6.10
Chick production and delivery	20	0	20	0.93
Others minor contributors	21	0	21	1.05
Rendering of mortalities (including transport)	-3	0	-3	-0.14
Total (excluding litter disposal)	1,686	434	2,120	100.00
Litter-to-power scenario	-219		-219	
Total (litter-to-power scenario)	1,467		1,901	
Litter-to-fertiliser scenario	-32		-32	
Total (litter-to-fertiliser scenario)	1,654		2,088	

[a] Others minor contributors: Shavings, Diesel, Chemicals, Water, Wash Water, Transport (LPG, diesel, shavings, chemicals, waste), Landfill

6.5.3 Life cycle energy use in broiler feed production and delivery

The fossil energy used in the production and delivery of broiler feed is presented in Figure 35. The total amount of fossil energy used in the production and delivery of one tonne of broiler feed was 4,197 and 4,192 MJ for the 'Standard' and 'Heavy' production systems respectively. The highest contributions to total fossil energy use were associated with inclusion of wheat, soya meal and field beans/rapeseed meal in broiler feed, which in total accounted for the 56 % (Figure 35).

Fossil energy consumption associated with the transport of ingredients to the feed mill and feed to the farms were also important contributors. The energy used in transport was the same for both the feed ingredients and the finished feeds because the mass of material and distance travelled was assumed to be similar. The energy used by the feed mill (heat and electricity) accounted for between 6.4 % and 7.1 % of the total fossil energy used in feed production.

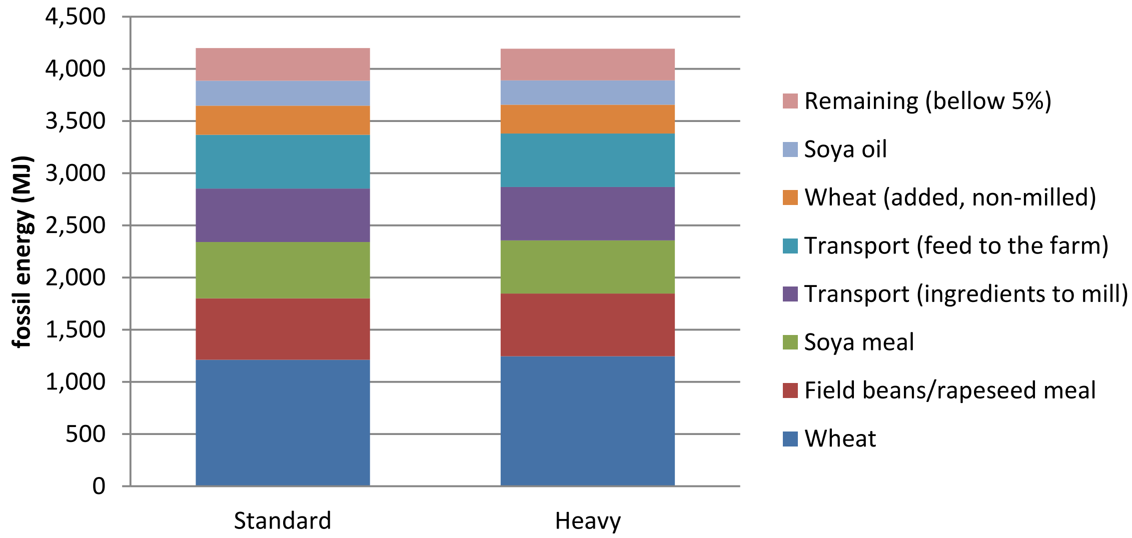


Figure 35 Fossil energy use associated with broiler feed production and delivery for the 'Standard' and 'Heavy' production systems in the UK (MJ/tonne)

6.5.4 Life cycle GHG emissions associated with broiler feed production and delivery

The GHG emissions associated with the production and delivery of broiler feed are presented in Figure 36. Total GHG emissions associated with the production of one tonne of broiler feed were 923 and 913 kg CO_2e, for the 'Standard' and 'Heavy' production systems respectively. These results include GHG emissions associated with land transformation, almost 100% of which were attributable to soya bean production in Brazil. Altogether, the inclusion of wheat, soya bean meal, field beans/rapeseed meal and soya oil in broiler feed accounted for 84 % of the total GHG emissions associated with feed production and delivery.

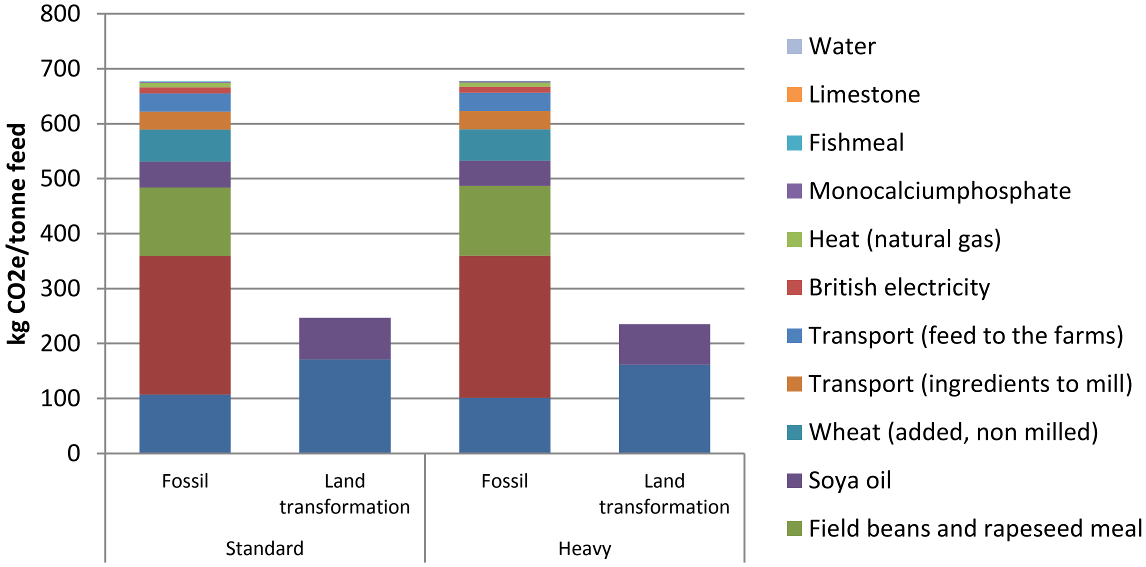


Figure 36 Greenhouse gas emissions associated with feed production and delivery for the 'Standard' and 'Heavy' broiler production systems (kg CO2e/tonne feed)

6.5.5 Sensitivity analysis

In the base case, the production of electricity by combustion of litter in fluidised bed combustion power plants displaces the production of average British electricity (highly coal based). If electricity had been produced from natural gas (considered marginal technology for UK electricity production) instead of average British electricity, fossil energy use in the 'Standard' production system, using the litter to power scenario, would have been 9,060 MJ/tonne live weight, which was 1% higher than the base case. GHG emissions would have been 1,858 kg CO_2e/tonne live weight which was 3% higher than the base case. The use of either average or marginal technology in the avoidance of electricity by the combustion of litter in the system modelled does not influence the results radically. This can be explained by the fact that both technologies are based on fossil fuels.

6.6 Discussion

6.6.1 Differences between broiler production systems

Fossil energy use and GHG emissions differed between the two production systems studies. The biggest differences being associated with FCR and on-farm energy use (Table 20). The FCR of birds on the 'Heavy' farms was 7% greater, than that of birds on the 'Standard' farms. Similarly, total fossil energy use (heat + electricity) was 10% higher on the 'Heavy' farms than that on the 'Standard' farms. As energy use for heating was similar for the two production systems. Differences in fossil energy use relate to differences in electricity use, with electricity use being 25% higher on the 'Heavy' than on the 'Standard' farms. Electricity is primarily used for ventilation and lighting. Differences in energy use can be attributed to the fact that the 'Heavy' birds took longer to grow and were less efficient than the 'Standard' birds. Consequently, to produce the same mass of birds, more electricity was used for ventilation and lighting. Differences in GHG emissions between the two production systems can be attributed to both on-farm electricity and feed use. Fossil energy use and GHG emissions/tonne of feed were similar for both the 'Heavy' and 'Standard' production systems. However, owing to differences in FCR more feed was used to produce the same mass of birds in the 'Heavy' than the 'Standard' production system.

6.6.2 Comparisons with other broiler production studies

Broiler production systems in the UK are fairly consistent, and the two systems studied represent 95% of UK broiler production (RSPCA, 2008). Williams *et al.* (2006) used LCA based on a systems modelling approach to study broiler production, and expressed the results on a dead weight basis. This model was developed further under DEFRA-funded project IS0222 and modified to express GHG emissions on a live weight basis for comparison with the current study. The results were 12,213 MJ and 2,016 kgCO_2e/tonne live weight for fossil energy use and GHG emissions respectively (Williams, pers. comm. 2011). In the current study, fossil energy use for the 'Standard' production system, using the litter to fuel scenario, was 27% lower and GHG emissions were 11% lower than those reported by Williams *et al.* (2006). However, their results are based on litter disposal to land as an organic fertiliser. When compared with the litter to fertiliser scenario in the current study, differences in energy and GHG emissions were only 9% and 2% respectively. Results obtained in the current study were based on data collected from farms to account for economic flow, and databases and calculations to provide 'static' life cycle inventories. In contrast, systems' modelling provides more 'dynamic' life cycle inventories which respond holistically to change. Although the approach adopted by Williams *et al.* (2006) was different to that adopted in the current study, the results are in good agreement.

Katajajuuri *et al.* (2008), cited in de Vries and de Boer (2010), provide results of 16,000 MJ and 2,079 kg CO_2e/tonne live weight for energy use and GHG emissions arising from broiler production in Finland, based on data collected from farms. These results are significantly higher than those obtained in the current study. In contrast, the life cycle inventory for broiler production for Denmark derived from the LCA food database (Nielsen *et al.*, 2003), included in the Simapro databases, provides results of 9,420 MJ and 1,820 kg CO_2e/tonne live weight, which are similar to those obtained in the current study. However, in the Danish products system modelled, significant credits are gained from the avoidance of rapeseed oil production through the co-production of soya oil when soya bean meal is produced. In the current study, system expansion is only used for disposal of wash water, litter and mortalities. Thus, these studies are not directly comparable.

Pelletier (2008) using data collected from farms, reported energy use and GHG emissions of 14,900 MJ and 1,395 kg CO_2e/tonne live weight respectively for the US broiler industry. For energy use and GHG emissions these results are higher and lower respectively than those obtained in the current study. Pelletier (2008) partitions 80% of energy use to feed production, with 18% being used on farm and 2% being attributed to chick production. In the current study, 62% of fossil energy was used for feed production and delivery, with 35% being used on farm and 3% for chick production. The percentage of energy used for feed production reported by Pelletier (2008) was considerably higher than that used in the current study. In addition, litter disposal accounted for -1,613 MJ/tonne live weight, which represents a lower credit than that of the litter-to-power scenario presented in the current study.

In the study of Pelletier (2008), feed production was associated with energy use and GHG emissions of 6,920 MJ and 612 kg CO_2e/tonne respectively. These values are 65% higher for energy use and 51% lower for GHG emissions than those obtained in the current study. Pelletier (2008), allocates GHG emissions to co-products based on gross energy. However, in the current study, ingredient life cycle inventories were derived from databases, such as the Ecoinvent database, which normally use economic allocation factors. As the approach to allocation adopted by the two studies was different, the results are not directly comparable. The use of economic allocation is probably more appropriate for feed production as the reason for processing is to provide a product to fulfil a human need in exchange for economic revenue. In the current study, the environmental burden associated with processing is ascribed to the co-products in proportion to their revenue as recommended (Guinee *et al.*, 2004; BSI, 2008b).

One of the major differences between the study reported by Pelletier (2008) and the current study relates to FCR. Pelletier (2008) reported a FCR of 1.9, which is 10% greater than that reported in the current study. If the FCR of birds in the 'Standard' production system using litter-to-power scenario had been similar then fossil energy use and GHG emissions would have been 9,724 MJ and 1,967 kgCO_2e respectively. Differences in FCR may be related to the economics of broiler production. A worse FCR may be accepted from a cheaper diet if overall profitability is increased. The aggregate diet reported by Pelletier (2008) consisted of 70% US corn, 20% US soya bean meal, 2.5% poultry by-product meal, 2.5% poultry fat, 2.5% US menhaden meal and 2.5% salt and limestone, which is significantly different to those presented in Table 21, with the main ingredient being corn instead of wheat. In addition, UK broiler diets contain a higher proportion of vegetable protein sources as the inclusion of terrestrial animal by-products in farm animal diets is currently prohibited under European legislation (EC, 2009). The GHG emissions for US corn reported by Pelletier (2008) were 328 kg CO_2e/tonne, which is mid-way between the values for the same product derived from the Ecoinvent databases (Ecoinvent Centre, 2010) and US Life Cycle Inventory (NREL, 2008) respectively. All of these values are considerable lower than the value of GHG emissions for wheat, derived from the Ecoinvent database and used in the current study. In addition, US soya bean meal production does not incur emissions associated with land transformation. The use of corn instead of wheat as the main dietary ingredient and US soya bean instead of Brazilian soya bean are probably the main reasons for lower GHG emissions reported for the US broiler industry.

6.6.3 GHG emissions of UK broiler production and mitigation options

The model used (Figure 32) has been developed as a way to represent typical broiler production facilities in the UK. The study has not included detailed assessment regarding the GHG emissions associated with breading stock farms. A simple test reveals that the GHG associated with broiler production could increase 10% if the effect of one generation of breeders is included.

Using values for broiler production in the UK (FAO, 2011) and GHG emissions calculated from the present study, GHG emissions from UK broiler production can be estimated to be 3.5 million tonnes/annum. This represents 0.6% of total UK GHG emissions for 2009 (DECC, 2011), although the emission associated with some crops (e.g. soya bean) do not occur in UK territory. The greatest opportunities for mitigation relate to feed production and utilisation. Potential mitigation strategies associated with crop production include different ways of improving the efficiency of fertiliser application such as precision farming techniques, use of slow release fertilisers, and timing of application (Smith *et al.*, 2008). The environmental impact of broiler production could also be reduced by diet formulations that reduce reliance on ingredients associated with relatively high GHG emissions, such as wheat, soya bean meal and field bean/rapeseed meal. For example, processed animal protein (PAP), derived from rendering of animal by-products, has a similar protein content to soya bean meal, but a significantly lower GHG emissions intensity (Chapter 4). In the current study, direct replacement of soya bean meal with mammalian PAP would reduce the GHG emissions associated with feed production by 26% and broiler production overall by 23%. However, as stated previously, the inclusion of terrestrial animal by-products in farm animal diets is currently prohibited under European legislation (EC, 2009). Reducing the FCR and maximising the efficiency of feed utilisation by appropriate diet formulation also offers a further opportunity for mitigation.

Reductions in on-farm energy use and litter management strategies may further reduce GHG emissions. Reduction in electricity use can be achieved by the implementation of more advanced control systems (e.g. variable frequency drive) for ventilation instead of ON-OFF controls (Teitel *et al.*, 2008). With regard to litter management, techniques such as reducing litter pH to inhibit the activity of microbes that convert organic matter to CH_4 have been discussed by Monteny *et al.* (2006). As with all life cycle approaches, it is important to realise that changes in one system may affect other systems. Consequently, all potential mitigation strategies would require careful evaluation before implementation. Changes in the energy sector (electricity and fuels production) towards major inclusion of renewable energy sources would also result in improvements in the poultry sector.

6.7 Conclusion

Fossil energy use and GHG emissions associated with two UK broiler production systems were determined using LCA methodology, using data collected from commercial farms. Fossil energy use and GHG emissions were 8,961 MJ and 1,798 kg CO_2e/tonne live weight for the 'Standard' production system, and 9,594 MJ and 1,901 kg CO_2e/tonne live weight for the 'Heavy' production system. The main contributors to energy use and GHG emissions were feed production, on-farm energy use and emissions associated with litter management. The results are in agreement with other studies. It is estimated that the UK broiler industry produces 3.5 million tonnes of CO_2/year, which represents 0.6% of total UK GHG emissions. The greatest mitigation opportunities relate to feed production and utilisation, on-farm energy use, and litter management strategies.

Chapter 7

The influence of co-product handling on the greenhouse gas emissions of processed livestock products: A case study of chicken derived co-products and by-products in the UK

7 The influence of co-product handling on the greenhouse gas emissions of processed livestock products: A case study of chicken derived co-products and by-products in the UK

7.1 Abstract

The study investigated the influence of co-product handling on the life cycle greenhouse gas (GHG) emissions of edible co-products (whole chickens, fillets, wings, etc) and poultry rendered products (poultry processed animal protein (PPAP), hydrolysed feather meal (HFM) and poultry rendered fat (PRF)). GHG emissions were calculated using economic flows from UK poultry processing industry. The co-product handling methods employed were economic and mass allocation, main product approach and system expansion. Economic flows of negative value were treated by system expansion. The influence of fuel type used in poultry by-product processing was also evaluated.

GHG emissions of mass weighted edible co-products were 3.381, 2.022, 3.460, and 3.057 kgCO_2e/kg for economic allocation, mass allocation, main product approach, and system expansion, respectively, when rendered fats were used as fuel in the poultry animal by-product (ABP) processing system. When economic allocation was used for poultry processing and mass allocation was used for poultry ABP processing GHG emissions of PPAP, HFM and PRF ranged between -0.166 and 1.195 kg CO_2e/kg. Different co-product handling combinations produced different life cycle impact assessment results. Economic allocation seems an appropriate method for the separation of co-products from by-products whereas mass allocation should be used following initial separation.

7.2 Keywords

co-product handling, allocation, system expansion, chicken, meat, by-product, rendering

7.3 Introduction

Life cycle assessment (LCA) has frequently been used to evaluate the environmental burden of animal products (de Vries and de Boer, 2010). However, most studies tend to concentrate on animal production, with system boundaries being the farm gate. Although, animal production is integral to the production of animal products, further processes are involved in the transformation of live animals to animal products. These post farm gate processes should be considered if the objective is to quantify the environmental burden of animal products. Several studies on complete meals, based on animal products, have included post farm gate processes such as slaughtering, meat processing and waste management (Sonesson *et al.*, 2005; Davis and Sonesson, 2008; Calderón *et al.*, 2010; Davis *et al.*, 2010). However, these processes have not generally been included in studies on the environmental impact of animal products (Williams *et al.*, 2006; Edward-Jones *et al.*, 2009). Animal slaughtering and meat processing plants produce a variety of co-products and by-products, and the inclusion of post farm gate processes in LCA studies requires careful examination of the different co-products, by-products and waste streams arising from each stage of production.

In LCA methodology a co-product is defined as any of two or more products arising from the same unit process or product system (ISO, 2006a). A by-product is defined as "a secondary product obtained during the manufacture of a principal commodity" (Meeker and Hamilton, 2006). A waste is defined as an economic flow with no or negative economic value (Guinee *et al.*, 2004). The production volume of a by-product is not dominated by its demand, but by the demand and production volume of the main product (or co-products).

Co-product handling methods are frequently debated in LCA methodology (Weidema, 1993; Azapagic and Clift, 1999b; Ekvall and Finnveden, 2001; Ayer *et al.*, 2007; Curran, 2007a; Reap *et al.*, 2008; Flysjö *et al.*, 2011a). However, it is commonly affirmed that there is no single method to solve the multiple output (or input) problem (Guinee *et al.*, 2004; Curran, 2007a; Kendall and Chang, 2009). The LCA standard (ISO, 2006a; b) provides a hierarchy for co-product handling and suggests that: 1) Allocation, defined as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" should be avoided, either by process division or system expansion, 2) If allocation has to be performed, the inputs and outputs of the system should be divided according to the physical relationships in which inputs and outputs change according to changes in their functions, 3) If no physical relations can be defined, other relationships should be used (e.g. economic value of products and co-products). Recently, LCA dedicated to assessing exclusively climate change have been developed (*Carbon Footprints* (CF)). The CF standardised by BSI (2008b) provides a similar hierarchy to that recommended by ISO (2006a, b). However, if allocation cannot be avoided, it should be based on the economic value of the products.

In practice, LCA/CF studies adopt one of two approaches to co-product handling; system expansion or economic allocation (Ayer *et al.*, 2007; Schau and Fet, 2008; Kendall and Chang, 2009). Allocation is normally used in attributional LCA studies, whereas system expansion is normally used in consequential LCA studies (Baumann and Tillman, 2004; Curran, 2007a; Schmidt, 2008b; Thomassen *et al.*, 2008a; Fruergaard *et al.*, 2009). The reason for this is that system expansion is adequate to analyse changes in the product system, demand or production volume (Ekvall and Finnveden, 2001; Cederberg and Stadig, 2003), but it requires more data to include avoided product systems (Ekvall and Finnveden, 2001; Reap *et al.*, 2008; Thomassen *et al.*, 2008a). Economic allocation is more generally applicable (Weidema, 1993; Guinee *et al.*, 2004), but economic value varies with time (Ayer *et al.*, 2007; Feitz *et al.*, 2007), tariffs and subsidies make it in-perfect (Feitz *et al.*, 2007; Schau and Fet, 2008) and it may not reflect the effect of decisions (Reap *et al.*, 2008). Choice of co-product handling method is one of the main reasons for variability in results and non-comparability between studies of similar products (Azapagic and Clift, 1999b; Cederberg and Stadig, 2003; Heijungs and Guinee, 2007; Reap *et al.*, 2008; Cherubini *et al.*, 2009; Flysjö *et al.*, 2011a).

Poultry meat from broiler chickens contributes 38% to UK meat production with 1.3 million tonnes per annum being produced between 2000 and 2008 (FAO, 2011). The importance of taking a holistic approach when studying the environmental burden of broiler production has been discussed by Pelletier (2008). In the UK, animal by-products (ABP) that arise from animal production and meat processing are categorised by EU legislation (EC, 2009) into category 1, 2 and 3 materials. Under this legislation, poultry by-products that are passed as fit for human consumption, but are not marketable, are classified as category 3 materials. These by-products have a price and are considered to be valuable commodities. Feathers are used to produce hydrolysed feather meal (HFM) and offal and bone are used to produce poultry processed animal protein (PPAP) and poultry rendered fat (PRF). Although classified as category 3 material, disposal of poultry blood represents a cost to poultry processors. Dead birds are classified as category 2 material, disposal of which also represents a cost to poultry processors. Category 2 material is normally mixed with and treated as category 1 material prior to disposal by rendering.

The influence of different co-product handling methods on the GHG emissions of ruminant co-products (milk and beef) has been studied (Cederberg and Stadig, 2003; Thomassen *et al.*, 2008a; Flysjö *et al.*, 2011a). However, in spite of being one of the most important animal products, the influence of different co-product handling methods on the GHG emissions of poultry meat co-products and by-products has not been investigated. The objectives of the study were to quantify the GHG emissions associated with different poultry meat co-products and by-products, and to investigate the influence of different co-product handling methods.

7.4 Materials and methods

7.4.1 System description and economic flows

The GHG emissions associated with UK broiler production to the farm gate, including emissions derived from fossil sources, agriculture and land transformation were derived from Chapter 6. Life cycle inventories for electricity production, heat from light fuel oil and natural gas, water and wastewater were taken from Ecoinvent databases (Ecoinvent Centre, 2010).

7.4.1.1 Poultry meat processing

Economic flows for poultry processing were provided for one UK plant by the VION Food Group Ltd (VION). Poultry processing involves two main stages. 1) Evisceration involves the slaughtering and separation of whole birds (co-products) from non-edible material (e.g. viscera, heads, feathers) and 2) Portioning involves the separation of eviscerated whole birds into different chicken parts (e.g. fillets, legs, wings, etc.), further processing of the whole chicken (including cooling). The plant studied received eviscerated whole birds for portioning both from its own evisceration facility and from external sources. Economic flows for poultry processing were calculated based on 1.0 kg broiler live weight (Table 26).

Table 26 Economic flows for the poultry processing plant during 2010

Poultry Co-products	**Unit**	**Amount**
Whole chickens (final)	kg	0.033
Chicken quarters, halves	kg	0.001
Wings	kg	0.062
Fillets	kg	0.209
Leg, drums, thighs	kg	0.197
Trims	kg	0.008
Edible offal	kg	0.042
Category 3 poultry by-products		
Offal + bone	kg	0.329
Feathers	kg	0.064
Inputs		
Live weight broiler	kg	1.000
Water	kg	2.439
Electricity	MJ	0.379
Heat (as energy content in Light fuel oil)	MJ	0.218
Transport (live birds from farm to plant)	kg-km	200
Negative economic flows		
Poultry blood	kg	0.030
birds dead on arrival, birds that do not pass post mortem inspection, heads, feet and floor waste	kg	0.024
Wastewater	m^3	0.002
Emissions to air		
Methane, chlorodifluoro-, HCFC-22	kg	1.155×10^{-5}

Poultry processing produces a large variety of edible co-products and poultry ABP. In the current study the edible co-products functional units employed were: 1 kg whole chicken, 1 kg fillet, 1 kg legs and thighs, 1 kg quarters and halves, 1 kg wings, 1 kg edible offal and 1 kg trims. In addition, only for process subdivision, three groups of edible co-products were also defined: 1 kg of whole chicken, 1 kg of fresh portions, and 1 kg of frozen portions. The poultry ABP functional units employed were 1 kg offal and bone, and 1 kg feathers. Edible offal consisted of gizzards, hearts, necks, hocks and liver. Packaging and further processing were not included in the system.

7.4.1.2 Category 3 poultry by-products processing

Economic flows for the processing of category 3 poultry ABP are presented in Table 27. Data was provided from one poultry ABP processing plant which processed 226,000 tonnes poultry ABP per year, of which 180,000 tonnes was offal and bone and 46,000 tonnes was feathers. The offal and bone throughput represented 30% of the category 3 poultry by-product processed in the UK. The plant produced three products. PPAP and PRF were produced from the rendering of offal and bone, whereas HFM was produced from the hydrolysis of feathers. However, energy use by the plant was integrated.

Table 27 Mean economic flows for the poultry by-product processing plant used in the study between 2006 and 2008, normalized for 1 kg of offal and bone

Economic flows	Unit	Amount
Poultry rendered and hydrolysed products		
PPAP	kg	0.163
PRF	kg	0.101
HFM	kg	0.067
Inputs		0.000
Offal + bone	kg	1.000
Feathers	kg	0.265
Electricity	kJ	224.3
Heat (total energy content in both fuels used)	kJ	2886.1
Water	kg	0.525
Sodium hypochlorite	kg	1.34×10^{-3}
Sodium hydroxide	kg	7.77×10^{-4}
Sulphuric acid	kg	3.69×10^{-4}
Chemicals (various)	kg	1.08×10^{-3}

PPAP = Poultry processed animal protein
PRF = Poultry rendered fat
HFM = Hydrolysed feather meal

The rendering industry uses both natural gas (fossil) and category 1 rendered fat (biogenic) as fuels to produce process heat. Two fuel scenarios were used in the current study. In scenario 1, 100% of process heat was derived from natural gas, whereas in scenario 2, 100% of process heat was derived from category 1 rendered fat. Process heat derived from category 1 rendered fat was calculated as described in Chapter 4. The functional units for which results are presented are 1 kg PPAP, 1 kg PRF and 1 kg HFM.

7.4.1.3 Category 2 poultry by-product processing

Category 2 poultry ABP consists of birds dead on arrival, birds that do not pass post mortem inspection, heads, feet and floor waste. These ABP have null or negative economic value and are classified as wastes, which are disposed of by rendering. Economic flows for the rendering of category 1 (or 2) ABP were derived from Chapter 4. In the system modelled (Figure 37) surplus category 1 (or 2) rendered fat produced by the system was used in the production of biodiesel using economic flows derived from (Lopez *et al.*, 2010). This avoided the production and use of fossil diesel. Use and production of fossil diesel life cycle inventories were derived from the Ecoinvent databases (Ecoinvent Centre, 2010).

Blood was disposed of in fluidized bed combustion (FBC) power plants to produce electricity. It was assumed that this avoided the production of average British electricity (coal based). Poultry blood was assumed to have a similar composition to the blood of other species with a dry matter (DM) content of 19% (Alberts *et al.*, 2008; The Franklin Institute, 2011) and a protein and fat content of 98.8 and 1.2% DM respectively (Paladines *et al.*, 1964). The Higher Heating Value was assumed to be 24.3 MJ/kg DM (Paladines *et al.*, 1964). The Hydrogen (H) content of protein and fat was assumed to be 6 % and 12 % by mass, respectively, and therefore the total H content in blood was estimated as 1.2 % DM. Calculations detailed by Quiroga *et al.* (2010) were used to determine a Lower Heating Value (LHV) of 4,191 kJ/kg fresh blood. The calculated LHV was used to estimate the amount of electricity produced from 1 kg of blood (754 kJ), assuming the blood is burnt in a FBC power plant to produce electricity with an efficiency of 18 % (Yassin *et al.*, 2009).

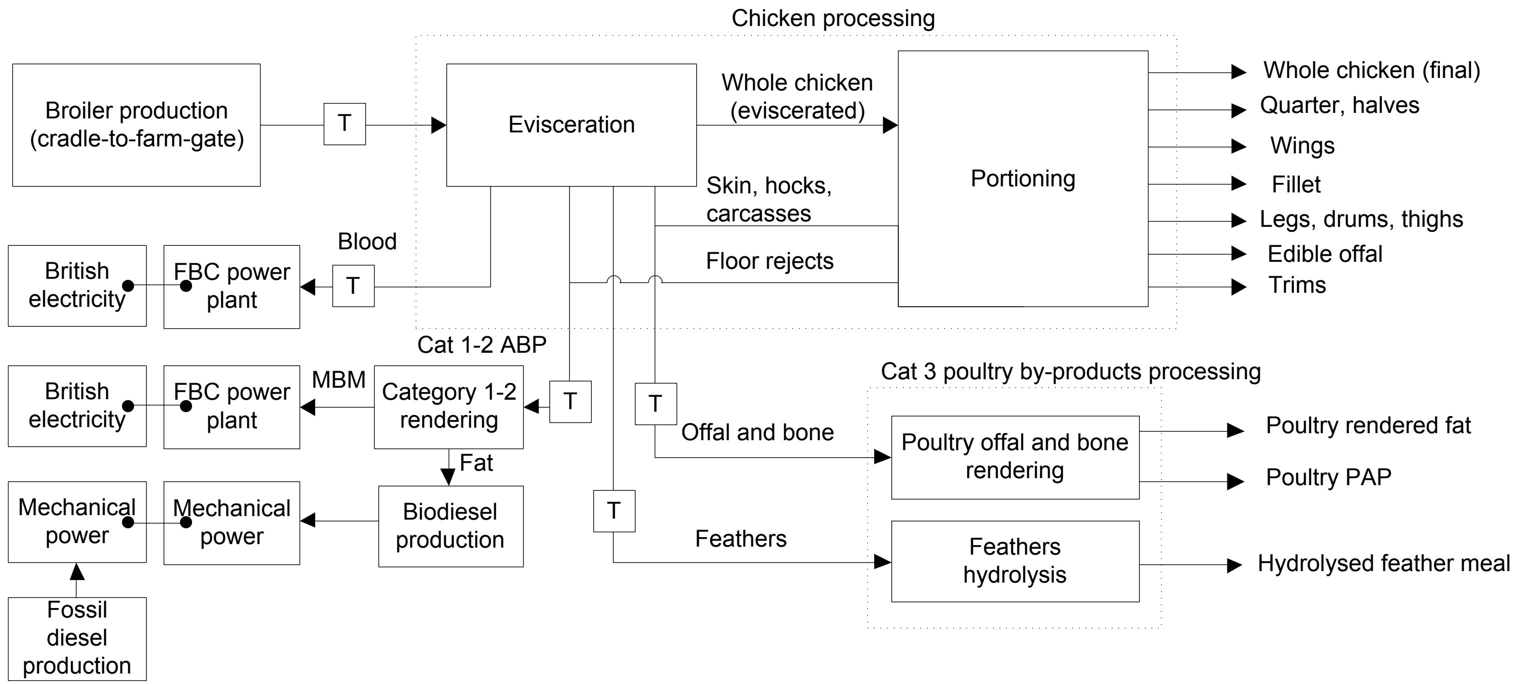


Figure 37 Poultry processing and poultry by-products processing schematic flow diagram (PAP: Processed Animal Protein, t: transport, ABP: animal by-product, FBC: fluidised bed combustion, MBM: meat and bone meal)

7.4.2 Co-product handling methods

In the system modelled, poultry blood and category 2 poultry ABP were treated as wastes which were disposed of by rendering or combustion in FBC power plants. The system was expanded to include the avoidance of fossil energy production. In addition, co-product or multiple input handling methods were required to allocate the environmental burden between different edible co-products and poultry ABP. Four methods were used in the study:

- Mass allocation (MA): the mass of the edible co-products and poultry ABP was used to divide the environmental burden from the previous stages and waste management.
- Economic allocation (EA): the mass weighted economic value of the edible co-products and poultry ABP was used to divide the environmental burden from the previous stages and waste management.
- Main product (MP): the edible co-products take the entire environmental burden from the previous stages and waste management.
- System expansion (SE): the system was expanded to include the avoidance of products from other products systems by the further processing of poultry ABP.
- Pseudo process subdivision: economic allocation in evisceration and partitioning according to electricity use in the portioning of three groups of products: whole chicken, fresh portions and frozen portions.

7.4.2.1 Poultry meat processing

Allocation factors for the edible co-products (and poultry ABP) of poultry meat processing are presented in Table 28. Economic allocation factors were derived from data produced by VION for the economic value for the different fractions. As the economic value of different fractions was confidential only the factors are presented.

Table 28 Allocation factors for edible co-products and poultry by-products arising from poultry processing

Edible co-products and poultry by-products	Mass allocation (%)	Economic allocation (%)	Main product approach / System expansion	
			Mass allocation (%)	Economic allocation (%)
whole chickens (final)	3.49	3.59	5.98	3.67
quarters, halves	0.15	0.22	0.25	0.22
wings	6.52	4.26	11.16	4.36
fillets	22.08	69.91	37.79	71.54
legs, drums, thighs	20.80	18.82	35.60	19.25
trims	0.90	0.41	1.54	0.42
Edible offal	4.49	0.53	7.69	0.54
Category 3 poultry by-products				
Offal + bone	34.79	2.26	0 / n.a.	0 / n.a.
Feathers	6.77	0.02	0 / n.a.	0 / n.a.

When system expansion was applied, poultry ABP processing was included within the product system of the edible co-products. It was assumed that PPAP and HFM replaced the production of soya bean meal, and that PRF displaced the production of soya oil, both produced from soya bean (Figure 38). The life cycle inventory for soya bean processing was taken from Nielsen *et al.* (2003). The inventory was modified to enable economic allocation factors to be derived using values taken from The World Bank (2011) and to use soya bean produced in Brazil. The life cycle inventory for soya bean processing presented by Nielsen *et al.* (2003) employed system expansion to include the avoidance of rapeseed oil production by the production of soya bean oil. In the current study, this system expansion was not appropriate as both co-products (soya bean meal and oil) are required. The life cycle inventory for the production of Brazilian soya bean was taken from the Ecoinvent databases (Ecoinvent Centre, 2010). When using system expansion, allocation was still required as there are various edible co-products. In this instance allocation was performed based on both mass and economics.

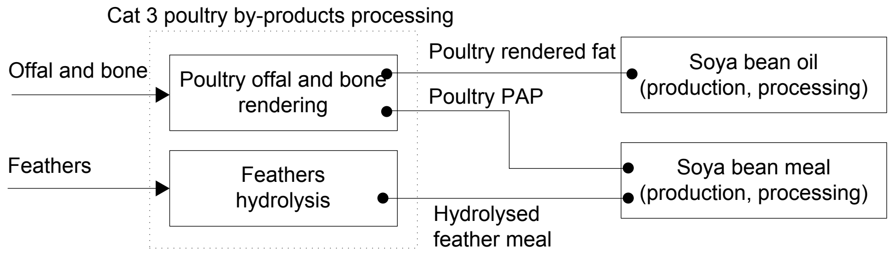


Figure 38 System expansion, processes avoided by processed category 3 poultry by-products (PAP: Processed Animal Protein)

Pseudo process subdivision results were obtained by economic allocation at evisceration and using process subdivision to divide the electricity use in portioning for three groups, whole chicken, fresh portions and frozen portions. Electricity use in portioning was 163, 191, and 715 kJ per kg of whole chicken, fresh portions, and frozen portions respectively. These were calculated as the electricity use intensity in the whole year for those three groups. Emissions of refrigerant fluid in portioning were ascribed according to the electricity use. All the other flows in portioning were ascribed according to the mass.

7.4.2.2 Category 3 by-products processing

The production of co-products from the processing of poultry ABP presents two problems. As data on economic inputs to the poultry processing plant was integrated (Table 29) it was difficult to allocate inputs to the two poultry ABP streams, offal + bone, and feathers. Similarly, the rendering of offal + bone produces two poultry rendered products, PPAP and PRF.

Multiple input mass allocation was performed based on the mass of offal + bone and feathers processed by the poultry ABP processing plant. Economic input allocation was performed based on the proportion of potential revenue from the rendering process and the potential revenue from the feather hydrolysis process. The potential revenue from the rendering process was calculated as the difference between the economic values of the PPAP and PRF produced and the offal + bone processed. Similarly, the potential revenue from the feather hydrolysis process was calculated as the difference between the economic value of the HFM produced and the feathers processed. Administrative and maintenance costs were not included. Allocation factors for poultry ABP processing (input allocation) and poultry rendered products are presented in Table 29.

Table 29 Allocation factors for poultry by-products processing (input) and poultry rendering products (output)

Input			**Output**		
Process	**Economic allocation (%)**	**Mass allocation (%)**	**Co-product**	**Economic allocation (%)**	**Mass allocation (%)**
Rendering	54.2	79.0	PPAP	59.55	61.65
			PRF	40.45	38.35
Hydrolysis	45.8	21.0	HFM	n.a. (100%)	n.a. (100%)

PPAP = Poultry processed animal protein
PRF = Poultry rendered fat
HFM = Hydrolysed feather meal

7.4.3 Calculation

The Simapro 7.3 ® software package (PRe Consultants, 2011) was used to model the products system. Climate change was assessed using the Greenhouse Gas Protocol 1.01 impact assessment method (The Greenhouse Gas Protocol, 2010). Only emissions that are considered to contribute to Climate change are reported (e.g. biogenic CO_2 emissions from the combustion of animal derived material such as MRF are not reported).

7.5 Results

The GHG emissions of different edible co-products and poultry ABP obtained using different co-product handling approaches are presented in Table 30. As stated, when system expansion was used poultry ABP (and the avoided production of substitutes) were included in the product system of the edible co-products. Similarly, when main product allocation was used, all the GHG emissions were allocated to the edible co-products of poultry processing. When economic allocation, mass allocation, and the main product approach were used, there was little difference in GHG emissions between the two rendering fuel scenarios for each edible co-product or poultry ABP produced. However, when system expansion was used the effect of fuel scenario on GHG emissions was slightly greater (Table 30).

Table 30 Greenhouse gas emissions of the edible co-products and by-products of the poultry processing (kg CO_2 / kg)

Co-products	RFS	Economic allocation	Mass allocation	Main product approach		System expansion	
				Economic allocation	Mass allocation	Economic allocation	Mass allocation
whole chickens	1	2.098	2.042	2.147	3.495	2.001	3.257
	2	2.100	2.044	2.149	3.498	1.901	3.095
fillets	1	6.465	2.042	6.616	3.495	6.165	3.257
	2	6.470	2.044	6.621	3.498	5.859	3.095
quarters, halves	1	3.015	2.042	3.086	3.495	2.876	3.257
	2	3.018	2.044	3.088	3.498	2.733	3.095
legs, drums, thighs	1	1.847	2.042	1.891	3.495	1.762	3.257
	2	1.849	2.044	1.892	3.498	1.674	3.095
wings	1	1.334	2.042	1.365	3.495	1.273	3.257
	2	1.335	2.044	1.367	3.498	1.209	3.095
trims	1	0.924	2.042	0.946	3.495	0.881	3.257
	2	0.925	2.044	0.947	3.498	0.838	3.095
edible offal	1	0.239	2.042	0.244	3.495	0.228	3.257
	2	0.239	2.044	0.245	3.498	0.216	3.095
Mass weighted edible co-product	1	3.415	2.042	3.495	3.495	3.257	3.257
	2	3.418	2.044	3.498	3.498	3.095	3.095
Category 3 by-products							
offal and bone	1	0.133	2.042	0.000	0.000		
	2	0.133	2.044	0.000	0.000		
feathers	1	0.005	2.042	0.000	0.000		
	2	0.005	2.044	0.000	0.000		

RFS: Rendering fuel scenario 1 = 100% natural gas and scenario 2 = 100% category 1 rendered fat

When mass allocation was used, GHG emissions of each edible co-product were equal for all the edible co-products and poultry ABP (2.042 kg CO_2/kg). However, when economic allocation was used the results ranged between 0.005 and 6.470 kg CO_2e/kg depending on their relative economic value, with fillets having the highest GHG emissions.

When system expansion or the main product approach were used during the first step of co-product handling, economic allocation resulted in different edible co-products having different GHG emissions, with the most valuable edible co-products having the highest and the least valuable having the lowest GHG emissions. For example, when natural gas was used as a fuel during rendering, GHG emissions for chicken fillets were 6.616 and 6.165 kg CO_2/kg fillet when the main product approach and system expansion were used respectively whereas, GHG emissions for edible offal were 0.244 and 0.216 kg CO_2/kg edible offal when the main product approach and system expansion were used respectively.

The mass weighted mean GHG emissions for all products illustrate the main effect of different co-product handling approaches for partitioning between edible meat and by-products. When mass allocation was used, edible co-products have the lowest GHG emissions because they are divided equally between all the edible co-products and poultry ABP. However, when the main product approach was used, edible co-products have the highest GHG emissions. When system expansion was used the GHG emissions associated with edible co-products were slightly lower than those produced using the main product approach. With system expansion edible co-products gain credits from the avoidance of soya bean meal and oil. The difference between the GHG emissions of edible co-products when either the main product approach or system expansion was used was 7 and 12% depending on the fuel scenario used. When economic allocation was used GHG emissions for edible co-product are mid-way between those of system expansion and the main product approach. No results can be obtained for poultry ABP when system expansion was used. Poultry ABP have null GHG emissions when the main product approach was used and low GHG emissions when economic allocation was used.

The contribution of broiler production (at the farm gate) to the GHG emissions of edible co-products and poultry ABP is presented in Table 31. Regardless of which rendering fuel scenario, broiler production accounted for approximately 93% of the GHG emissions associated with production of each edible co-product or poultry ABP when employing main product or allocation. When system expansion was adopted the contribution of broiler production was different depending on the fuel scenario (100 and 105% for fuel scenario 1 and 2). A contribution of greater than 100% is explained by the fact that the poultry processing system gains credits from avoidance of the production of soya bean meal and soya oil.

Table 31 The contribution of on farm broiler production (%) to the greenhouse gas emissions of edible co-products and poultry by-products

Rendering fuel scenario	Economic allocation	Mass allocation	Main product approach		System expansion	
			Economic allocation	Mass allocation	Economic allocation	Mass allocation
1	93.1%	93.1%	93.1%	93.1%	99.9%	99.9%
2	93.0%	93.0%	93.0%	93.0%	105.1%	105.1%

Rendering fuel scenario 1 = 100% natural gas and scenario 2 = 100% category 1 rendered fat

The GHG emissions obtained with pseudo process subdivision indicating the contribution from animal production are presented in Table 32. Under this co-product handling approach, the results ranged between 2.672 and 2.779 kg CO_2 per kg co-product group. The lowest relative contribution from broiler production (90.2%) and the highest GHG emissions were calculated for the frozen portions.

Table 32 Greenhouse gas emissions (kg CO_2/kg) and contribution of on farm broiler production (%) according to pseudo process subdivision for whole chicken, fresh portions and frozen portions

	whole chicken		fresh portions		frozen portions	
RFS	1	2	1	2	1	2
GHG (kg CO_2 / kg)	2.672	2.674	2.686	2.689	2.776	2.779
broiler production contribution (%)	93.8	93.7	93.3	93.2	90.2	90.2

RFS: Rendering fuel scenario 1 = 100% natural gas and scenario 2 = 100% category 1 rendered fat

The GHG emissions associated with production of poultry rendered products are presented in Table 33. The results vary considerably depending on the co-product handing approach adopted for poultry processing, the allocation approach adopted for poultry ABP processing and the rendering fuel scenario used. The GHG emissions of PPAP were highest (8.423 kg CO_2/kg PPAP) when mass allocation was used for both poultry processing and poultry ABP processing, and natural gas was used as a fuel during rendering. However, they were lowest (-0.099 kg CO_2/kg PPAP) when the main product approach was adopted for poultry processing, economic allocation was used for poultry

ABP processing, and category 1 rendered fat was used as a fuel during rendering. The GHG emissions of PRF and HFM were highest (8.672 and 9.587 kg CO_2/kg respectively) when mass allocation was used for poultry processing, economic allocation was used poultry ABP processing and natural gas was used as a fuel during rendering. However, the lowest GHG emissions for PRF (-0.178 kg CO_2/kg PRF) were obtained when the main product approach was adopted for poultry processing, mass allocation was adopted for poultry ABP processing and category 1 rendered fat was used as a fuel in rendering. For HFM the lowest GHG emissions (-0.489 kg CO_2/kg HFM) were obtained when main product approach was used for poultry processing, economic allocation was used for poultry ABP processing and category 1 rendered fat was used a fuel for rendering.

Table 33 Greenhouse gas emissions of poultry rendered products (kg CO_2/kg)

Poultry processing co-product handling method		**Economic allocation**		**Mass allocation**		**Main product approach**	
Poultry by-products processing co-products handling method		**Economic allocation**	**Mass allocation**	**Economic allocation**	**Mass allocation**	**Economic allocation**	**Mass allocation**
	RFS						
PPAP	1	0.967	1.201	7.943	8.423	0.482	0.698
	2	0.387	0.325	7.369	7.555	-0.099	-0.178
PRF	1	1.056	1.201	8.672	8.422	0.526	0.698
	2	0.422	0.325	8.046	7.554	-0.108	-0.178
HFM	1	1.552	0.758	9.675	8.881	1.531	0.736
	2	-0.468	-0.166	7.662	7.966	-0.489	-0.188

PPAP = Poultry processed animal protein
PRF = Poultry rendered fat
HFM = Hydrolyzed feather meal
RFS: Rendering fuel scenario 1 = 100% natural gas and scenario 2 = 100% category 1 rendered fat

7.6 Discussion

7.6.1 Data

The discussion of data is limited to energy use during poultry processing and poultry ABP processing. For both of these processes energy use was the main contributor to GHG emissions. The energy use and GHG emissions arising from broiler production are presented in Chapter 6 and constitute the most important contribution to GHG emissions associated with production of both edible co-products and poultry ABP.

In the processing plant studied the relative yield of edible co-products and poultry ABP was 55% and 39% respectively. However, it is important to realise that the plant studied produced both eviscerated whole chickens and chicken portions. The yield of edible co-product would be increased and yield of poultry ABP reduced if the plant only produced eviscerated whole chicken.

During the evisceration stage, process heat and electricity use was 169 kJ and 228 kJ/kg broiler live weight, whereas, during the portioning stage heat and electricity use was 71 kJ and 214 kJ/kg eviscerated whole chicken. The electricity and heat use is similar to other facilities of the same company. With a yield of 0.702 kg eviscerated whole chicken per kg broiler live-weight the total process heat and electricity used during broiler processing was 218 kJ and 379 kJ/kg broiler live-weight respectively (Table 26). Hence, the evisceration stage accounted for 77% and 60% of the process heat and electricity use during poultry processing. Data on energy use during poultry processing in the literature is limited. Energy use for poultry slaughtering in Finland has been reported to be 792 kJ and between 1728-1764 kJ/kg slaughtered bird for process heat and electricity use respectively (The Finnish Environment, 2002). Similarly, energy use for poultry slaughtering reported in the Danish LCA food databases is 360 kJ and 720 kJ/kg live-weight for process heat and electricity respectively (Nielsen *et al.*, 2003). Energy use in the current case study was comparatively low in relation to other published studies. The electricity use in portioning according to process subdivision was 163, 191, and 715 kJ per kg of whole chicken, fresh portions, and frozen portions respectively. This indicates that depending on the proportion of fresh or frozen products produced the average electricity use intensity can be considerably from plant to plant.

The data on poultry ABP processing was provided by one plant that processed both offal and bone (rendering) and feathers (hydrolysis). The heat production of this plant was integrated therefore discussion of the data is based on the total mass of poultry ABP processed. Process heat and electricity use was 2281 kJ and 177 kJ/kg poultry ABP processed respectively. During ABP processing, thermal energy is used primarily to evaporate water. In the current study, the yield of poultry rendered products was 0.26 kg/kg poultry ABP processed (Table 27). Energy use for the rendering of mammalian ABP has been reported in Chapter 4, with thermal energy use being 1357 kJ and 2646 kJ, and electricity use being 260 kJ and 375 kJ/kg category 3 and category 1 ABP processed respectively. These values correspond to rendered product yields of 0.57 kg and 0.40 kg/kg ABP processed, for category 3 and 1 ABP respectively. In the current study, thermal energy use was midway between the values reported for category 3 and 1 mammalian ABP. However, the amount of water removed by evaporation was higher. Data on energy used during poultry ABP processing in the US has been reported by Lopez et al. (2010). Process heat and electricity use were 2439 kJ and 230 kJ/kg offal and bone respectively with a rendered product yield of 0.40 kg/kg poultry ABP processed. Again thermal energy use was similar, but rendered product yields were higher than those reported in the current study. This may be related to the greater integration of the plant for which data is reported here. The energy use in poultry ABP rendering is in good agreement with values in the literature and in particular with energy use in mammalian ABP rendering in the UK.

7.6.2 Co-product handling approaches

In the current study the GHG emissions of edible co-products differed greatly depending on the co-product handling approach adopted. When the main product approach was used, GHG emissions for the mass weighted co-products were 71% higher than those obtained using mass allocation. In the case of chicken fillets, the highest GHG emissions were obtained using the main product approach with economic allocation and the lowest using mass allocation. In the case of poultry rendered products, GHG emissions varied greatly depending on the co-product handling approach adopted during poultry processing.

In the UK, ABP are defined by legislation (EC, 2009). However, in this paper the criterion as to whether a by-product is treated as a co-product or a waste is whether or not it had a positive or negative economic value (Guinee *et al.*, 2004). For the purposes of LCA methodology, ABP with negative economic value are treated as wastes. System expansion is always used for disposal of wastes (economic flows with negative value). In the system studied, poultry blood was considered a waste and was destroyed with energy recovery in a FBC power plant. However, poultry blood is classified under EU legislation as category 3 materials (EC, 2009), which could potentially be processed to produce poultry blood meal. In this scenario, poultry blood may attract a positive economic value and would have to be treated as an additional poultry ABP.

Economic allocation approach appears to be well suited to poultry processing, where GHG emissions for both edible co-products and poultry ABP are required. Economic allocation is justified by the fact that the purpose of poultry production and processing is to meet a demand for poultry meat. However, some of the environmental burden should also be attributed to category 3 poultry ABP that have a positive economic value. At the other extreme, using the main product approach, the entire environmental burden is attributed to the edible co-products. This approach was adopted in Chapter 4 for beef derived ABP, the disposal of which represented a cost to meat producers in the UK.

When system expansion was used, GHG emissions were only obtained for edible co-products. Poultry rendered products are used mainly to replace marginal protein and fat sources, in this case Brazilian soya bean (including agricultural and land transformation emissions). The avoidance of soya bean meal and soya oil reduced the GHG emissions of edible co-products in comparison to the main product approach or economic allocation. This is similar to the system expansion for milk production presented by Flysjö *et al.* (2011a). The extent of the reduction in GHG emissions also varied slightly depending on the type of fuel used in poultry ABP processing.

Arguably, when the objective of the study is to calculate the potential environmental impact of food production systems, system expansion is an appropriate approach to use, as the system is expanded to include the effects associated with disposal of unavoidable by-products. System expansion included the replacement of Brazilian soya bean meal and soya oil. The results may have been slightly different if soya bean meal and soya oil had been sourced from other parts of the world, or if other protein and fat sources had been used. In the study of soya bean processing used (Nielsen *et al.*, 2003), soya bean meal and soya oil are produced at a ratio of 4.6:1. In the current study, the production ratio of poultry protein meals (PPAP+HFM) to PRF was 2.8:1. The requirement for additional marginal oil was not included in the system modelled because analysis of the consequential loops arising from this system was not part of the scope of the study. Consequential loops for soya bean meal have been studied by Dalgaard *et al.* (2008). However, if the marginal oil source used had been palm oil from Malaysia instead of soya oil, the GHG emissions would have increased by 3-4 %, suggesting the marginal oil used had a minor (but not negligible) effect on the GHG emissions of edible co-products (in the current case study). If palm oil had been chosen only to complete the avoidance of fat by the production of PRF by the extended poultry meat system, the change in the GHG emission of the edible products would have been lower than 3-4%.

When main product, economic allocation or mass allocation were used, GHG emissions for edible co-products were slightly lower when natural gas was used as a fuel for poultry ABP processing. This is because in the system modelled, surplus category 1 (and 2) rendered fat was used for the production of biodiesel, which avoided the production and use of fossil diesel (Figure 37). More credits were gained from the use of rendered fat to replace fossil diesel, than were gained from the use of rendered fat to replace natural gas. However, when system expansion was used, GHG emissions for the edible products were higher when natural gas was used as a fuel. Although this seems counter intuitive, this occurred principally because the amounts and destinies of different category rendered products are different. When employing system expansion, additional credits for offset of fossil fuels are gained when category 1 rendered fats are used as fuel (Chapter 4).

As the production of both edible co-products and poultry ABP is dependent on the demand for poultry meat, the demand for poultry ABP does not influence their production volume. Therefore economic allocation seems to be an appropriate approach to allocate GHG emissions between edible co-products and poultry ABP. However, the production of different edible co-products may be influenced by demand; therefore mass allocation may be more appropriate at this stage (the mass weighted result for all edible co-products).

The difference in GHG emission for poultry rendered products using different co-product handling combinations is considerable, and mainly associated with the contribution from broiler production. For example, when mass allocation was used during poultry processing, GHG emissions for poultry rendered products were higher than those of edible co-products. When mass allocation was used a significant proportion of the environmental burden associated with broiler production was attributed to the poultry rendered products. During rendering or hydrolysis, poultry rendered products are dried. Therefore, a higher mass of birds is required to produce 1.0 kg of poultry rendered product than would be required to produce 1.0 kg of poultry fillet. There are also additional GHG emissions associated with the use of natural gas as a fuel during poultry ABP processing. Alternatively, when the main product approach or economic allocation was used during poultry processing, GHG emissions for poultry rendered products were significantly lower, and in some cases negative results were obtained when category 1 (and 2) rendered fat was used as a fuel (associated with the credits of the Category 1 system). As the production of different poultry rendered products (PPAP, MRF, HFM) is dependent of poultry ABP supply and not demand driven, mass allocation would seem to be an appropriate approach to allocate GHG emission between different poultry rendered products.

Process subdivision is highest in the hierarchy of co-product handling choices according to both BSI (2008b) and ISO (2006a, b). In the current study, a pseudo process subdivision method was used to divide between whole chicken, fresh portions, and frozen portions. For example, there are fillets, wing, legs that either leave the plant fresh or are cold stored to be sold later in the year. The electricity use in portioning was 3.7 times higher in the case of frozen portions. However, the GHG emissions associated with frozen portions were only 3% higher than those of the fresh portions, mainly because broiler production is the most important contributor. Process subdivision is well suited to account for the differences in processing.

7.6.3 Greenhouse gas emissions of poultry rendered products

Category 3 poultry rendered products are mainly used as protein and fat sources in pet foods. The GHG emissions associated with mammalian rendered products have been reported in Chapter 4. In the UK disposal of mammalian ABP represents a cost to the producer and therefore for LCA purposes they were considered to be wastes, which do not carry any of the environmental burden associated with their production. In the current study, poultry ABP had a positive economic value and were therefore considered to be valuable co-products which should carry some of the environmental burden associated with their production. When mass allocation was used to allocate the environmental burden between category 3 mammalian processed animal protein (PAP) and mammalian rendered fat (MRF), GHG emissions for PAP or MRF ranged from 0.05 to 0.29 kg CO_2e/kg depending on the relative proportions of natural gas and MRF used as fuels during the rendering process (Chapter 4). Assuming economic allocation is used during poultry processing and mass allocation is used during poultry ABP processing, GHG emissions for PPAP or PRF ranged between 0.325 and 1.201 kg CO_2/kg depending on the fuel used during poultry ABP processing. The higher values are attributable to the fact that poultry rendered products carry some of the environmental burden associated with their production. If the main product approach was used during poultry processing, and poultry ABP were considered to be wastes similar to mammalian ABP, GHG emissions for PPAP or PRF ranged between -0.178 and 0.698 kg CO_2e/kg. On average between 2006 and 2008, the UK rendering industry derived 75% of its thermal energy from category 1 MRF and 25% from natural gas. Using this scenario, GHG emissions for mammalian PAP and MRF were 0.15 and 0.15 kg CO_2/kg respectively (Chapter 4). Assuming economic allocation was used for poultry processing and mass allocation was used for poultry ABP processing, GHG emissions for PPAP or PRF and HFM were calculated to be 0.544 and 0.065 kg CO_2/kg respectively.

Dalgaard *et al.* (2008) reported consequential GHG emissions for soya bean meal of 0.721 and 0.344 kg CO_2e/kg depending on whether palm or rapeseed oil were used as marginal oil sources respectively. The same study reported attributional GHG emissions of 0.726 and 0.901 kg CO_2/kg depending on whether economic or mass allocation was used. The GHG emissions associated with land transformations were not included. Their results included oceanic transport; however this is a minor contributor in relation to agricultural associated GHG emissions. In the current study, the value used for soya bean (Ecoinvent Centre, 2010) included GHG emissions associated with land transformations and was considerably higher than those reported by Dalgaard *et al.* (2008). The GHG emissions associated with PPAP and HFM (when produced with 75% rendered fats) are lower than those reported for soya bean meal by Dalgaard *et al.* (2008) and significantly lower than those for Brazilian soya bean meal calculated from the Ecoinvent databases (Ecoinvent Centre, 2010). The main reason for this difference reflects the fact that the objective of soya bean production is to produce soya bean meal. Consequently, they carry the environmental burden associated with agricultural production (and land transformation). If economic allocation is used during poultry processing the contribution of broiler production to the GHG emissions of poultry rendered products is relatively low. In addition, when category 1 rendered fat is used as fuel for poultry ABP processing, additional credits are provided to the poultry ABP processing system.

7.7 Conclusions

The use of different co-product handling approaches during poultry processing and poultry ABP processing produces different GHG emissions for different edible co-products and poultry rendered products. Allocation is essential in attributional studies where results are required for all the co-products and by-products produced. As the main purpose of meat production systems is to produce meat, economic allocation should be used to allocate

GHG emissions between the edible co-products and by-products. However, as no co-product or by-product is considered to be more desirable than another, process subdivision or mass allocation should be used following initial separation. When the objective of the study is to investigate the environmental burden of the whole food system, the system should be expanded to include the effects associated with disposal of unavoidable by-products.

Provided economic allocation is used during poultry processing to allocate GHG emissions between edible co-products and by-products, the GHG emissions of poultry rendered products such as PPAP, PRF and HFM are lower than those of alternatives such as soya bean meal and soya oil. Using this co-product handling approach, the GHG emissions of poultry rendered products depends largely on the relative proportion of category 1 rendered fat and natural gas used as fuels during poultry ABP processing.

Chapter 8

The effect of inclusion of ingredients derived from terrestrial animal by-product on the greenhouse gas emissions of animal production: A case study of salmonid feed production in UK

8 The effect of inclusion of ingredients derived from terrestrial animal by-product on the greenhouse gas emissions of animal production: A case study of salmonid feed production in UK

8.1 Abstract

The influence of the inclusion of terrestrial animal derived by-products in salmonid diets was analysed from a Climate Change perspective using LCA methodology. Four formulations were evaluated: conventional (based on fish and vegetable derived ingredients), maximised fish meal, maximised feather hydrolysed meal plus porcine haemoglobin meal, and maximised poultry by-product meal. Fuel use in the processing of poultry derived ingredients was evaluated with rendered fats and natural gas. Two sources of fish derived ingredients were evaluated: white fish by-products and small pelagic fisheries. Three co-product handling combinations were used: economic and mass allocation, and an additionally defined combination.

Results indicated that when using mass allocation the inclusion of animal by-product derived ingredients results in higher GHG emissions than the conventional formulation. In contrast results based on economic allocation indicated that the inclusion of animal by-product derived ingredients results in lower GHG emissions than conventional or maximised fish meal formulations. It is argued that economic allocation is adequate for animal by-product systems, as they are unavoidable by-products of meat production, whose demand does not influence their production volume.

8.2 Keywords

Salmon, feed, co-product, by-product, life cycle assessment, carbon footprint

8.3 Introduction

The environmental impact of different food production systems has increasingly been studied in recent years, in particular employing quantitative methodologies such as life cycle assessment (LCA) (Pelletier *et al.*, 2007; Roy *et al.*, 2009; Calderón *et al.*, 2010; de Vries and de Boer, 2010; Cerutti *et al.*, 2011; Henriksson *et al.*, 2011; Milani *et al.*, 2011).

Generic LCA, standardised by ISO standards (ISO, 2006b; a) can be used to evaluate different environmental impacts. Greater global awareness of the impacts of Climate Change has led to further LCA specification towards the study of only greenhouse gas (GHG) emissions; such studies may be referred to as *Carbon Footprints* (BSI, 2008b).

Land based animal production is one the most important contributors to Climate Change, accounting for 18% of global GHG emissions (Steinfeld *et al.*, 2006). In contrast, fisheries and aquaculture are regarded as a relatively minor contributor to global GHG emissions (FAO, 2008b). However aquaculture is the animal food production sector with the highest growth rate. Globally farmed salmon, trout and smelts production increased from 1.4 to 2.3 million tonnes between 1999 and 2008 (FAO, 2008a); an increase of 64%. This trend is likely to continue, as one half of the wild fish stocks (for which information is available) have been fully exploited, and therefore the increasing demand for fish in the future will have to be met by aquaculture (FAO, 2008b; Cressey, 2009). Total land based animal meat production in 2008 was 279 million tonnes (FAO, 2011). Salmonids are among the most heavily farmed aquatic species in the Western world and although salmonid farming is not currently a major meat producing sector, the rapid development of this sector has driven interest in quantifying its environmental burden.

There have been several LCA studies on salmonid farming systems (Ellingsen and Aanondsen, 2006; Gronroos *et al.*, 2006; Aubin *et al.*, 2009; Ayer and Tyedmers, 2009; d'Orbcastel *et al.*, 2009; Ellingsen *et al.*, 2009; Pelletier *et al.*, 2009; Boissy *et al.*, 2011) and in general of aquaculture systems (Henriksson *et al.*, 2011). Pelletier *et al.* (2009) found the average GHG emissions of salmon produced in Norway, the UK, Canada and Chile were 1.790, 3.270, 2.370 and 2.300 kgCO_2e/kg live weight, respectively. Ayer and Tyedmers (2009) found the GHG emissions of salmonid farming systems in Canada were 2.073, 1.900, 2.770 and 28.200 kgCO_2e/kg live weight for conventional marine net-pen system, marine floating bag system, land-based saltwater flow-through system, and land based freshwater recirculating system, respectively (the latter used significantly higher amounts of electricity and is regarded as a niche technology). Aubin *et al.* (2009)

calculated GHG emissions of 2.753 $kgCO_2e/kg$ live weight rainbow trout cultured in flow-through systems in France. The GHG emissions of conventional salmonid production are similar or slightly higher than those of broiler chicken production (1.798 and 1.395 $kgCO_2e/kg$ live weight for UK (Chapter 6) and US (Pelletier, 2008) production, respectively). It should be noted that broiler chicken production is regarded as the most efficient animal protein production system (Pelletier, 2008). Relatively low GHG emissions intensity and low production volume when compared to main land based meat production systems indicates that salmonid farming is arguably not a major contributor to Climate Change as yet.

There is general agreement that feed production is the most important contributor to Climate Change in the life cycle of farmed salmonids (aside from the highly energy intensive land based recirculating systems in which the relative contribution of feed production is diluted by the electricity used but still is similar in magnitude to that in other systems). Pelletier *et al.* (2009), Aubin *et al.* (2009), and d'Orbcastel *et al.* (2009) quantified the relative contribution from feed production as 94%, 73% and between 88% and 91%, respectively. This indicates that the GHG emission intensity of farmed salmonid systems is very sensitive to their Feed Conversion Ratio (FCR) and the GHG emissions intensity of the feed production system. Again this is similar to findings for broiler chicken production by Pelletier (2008) and Chapter 6.

Animal by-products are secondary products of the animal production chain, whose allowed uses are specified in EU legislation (EC, 2009). Under this legislation, poultry and porcine by-products (i.e. feathers, offal and bone, porcine blood) are classed as category 3 materials that can be used as pet food ingredients but must not enter the human food chain (including farmed animals feeds). In contrast fish by-products (fish trimmings) can be used in farmed animals feeds (both aquatic and terrestrial non-ruminant). Globally, it has been estimated that in 2006 aquaculture used 68% and 89% of the fish meal and oil produced, respectively (Tacon and Metian, 2008). Sources of fish biomass to produce fish meal and oil vary significantly from country to country (Peron *et al.*, 2010). Pelletier *et al.*

(2009) presented annual average formulations for farmed salmonid feeds for 4 important salmon producing countries, which varied radically in source of fish biomass. Globally, the main source of fish biomass for the production of fish meal and oil are different small pelagic species caught for this purpose (Peron *et al.*, 2010).

From a Climate Change perspective, the inclusion of animal by-products as ingredients in salmonid feeds has been a subject of disagreement (Papatryphon *et al.*, 2004; Pelletier and Tyedmers, 2007; Pelletier *et al.*, 2009). Pelletier and Tyedmers (2007) and Pelletier *et al.* (2009) recommended that salmonid feed formulations should avoid animal by-products (both aquatic and land based) and recommended that inclusion of dedicated vegetable sources of fats and proteins should be maximised. Pelletier and Tyedmers (2007) indicated that the environmental impacts are much larger when using fish biomass from by-products from highly energy intensive fisheries in the production of fish meal and oil than from small pelagic species from energy efficient dedicated fisheries (reduction fisheries). The authors stated that this is contrary to the prescriptions of major organic aquaculture standards and previous findings by Papatryphon *et al.* (2004). Papatryphon *et al.* (2004) stated that the use of fish biomass from by-products results in a lower Climate Change impact than that from dedicated fisheries. Ellingsen and Aanondsen (2006) recommended the use of vegetable protein and fat sources instead of fish derived ingredients in the feeding of salmon. Boissy *et al.* (2011) found that the partial replacement of vegetable biomass instead of small dedicated pelagic fish biomass resulted in a 6% decrease in the GHG emissions associated with trout feeds; however it resulted in an 18% increase in salmon feeds. The difference was related to the GHG intensity of the production of the substitute vegetable feed ingredients used in each feed (not the same for trout and salmon). Neither Ellingsen and Aanondsen (2006) nor Boissy *et al.* (2011) included ingredients derived from terrestrial animal by-products in their evaluations.

In comparison to Papatryphon *et al.* (2004), Pelletier and Tyedmers (2007) suggested the reason for their radically different results was due to the co-product handling approach used. Papatryphon *et al.* (2004) used allocation based on the economic value of co-products and by-products whilst Pelletier and Tyedmers (2007) and Pelletier *et al.* (2009) used allocation based on gross energy content as recommended by Ayer *et al.* (2007). Ayer *et al.* (2007) reviewed different co-product handling strategies used in the LCA of seafood products (fisheries and aquaculture) and found that economic allocation is the most common approach. However they stated that this approach seems arbitrary and proposed that a better approach would be the use of the gross energy content in the co-products as they argued that it represents in a more realistic manner the biophysical flows for alternative feed productions strategies. Pelletier and Tyedmers (2007) also demonstrated that life cycle impact assessment results are very similar using either gross energy content or mass allocation. It is well documented, that different co-product handling methods is one of the reasons for different results and non-comparability among LCA studies of similar products (Azapagic and Clift, 1999b; Cederberg and Stadig, 2003; Heijungs and Guinee, 2007; Reap *et al.*, 2008; Thomassen *et al.*, 2008a; Cherubini *et al.*, 2009; Flysjö *et al.*, 2011a).

It seems that the inclusion of animal by-products derived ingredients in salmonid feeds provides either Climate benefits or costs depending on the co-product handling approach employed. However, the GHG emission intensity of terrestrial animal by-product derived ingredients production has not been taken into account properly in previous studies. The terrestrial animal by-product processing industry (in the UK) produces meal and proteins using different fractions of natural gas and category 1 mammalian rendered fat (a biofuel) for their thermal requirements (Chapter 4). It has been shown that terrestrial animal by-product derived proteins and fats have relatively low GHG emissions provided: (i) they have low value or no value at all and therefore under the used methodological choices regarding co-product handling the contribution from their production is very low or null, and (ii) in their processing they offset the production of fossil energy (Chapters 4 and 7).

The main objective of the current study was to compare the GHG emissions associated with the production of four alternative salmonid feed formulations in the UK (one conventional, one with maximised inclusion of fish meal, one with maximised inclusion of poultry by-product meal and one with maximised inclusion of feathers hydrolysed meal plus porcine haemoglobin meal) under different co-product handling combinations taking into consideration the type of fuel employed in the UK terrestrial animal by-product processing industry. Two alternative sources of fish biomass were evaluated: fish by-product (from white fish fisheries for human consumption) and a small pelagic species from reduction fisheries.

8.4 Materials and Methods

8.4.1 System description, calculation and data sources

Figure 39 illustrates the salmonid feed production system modelled in this study and Table 34 presents the salmonid feed formulations evaluated, which were provided by a UK salmonid feeds production company. All four feeds were formulated to provide the same nutritional specification using a least cost ration formulation software package. Vitamins and other supplements were excluded from the analysis as their inclusion rates were very low (Table 34).

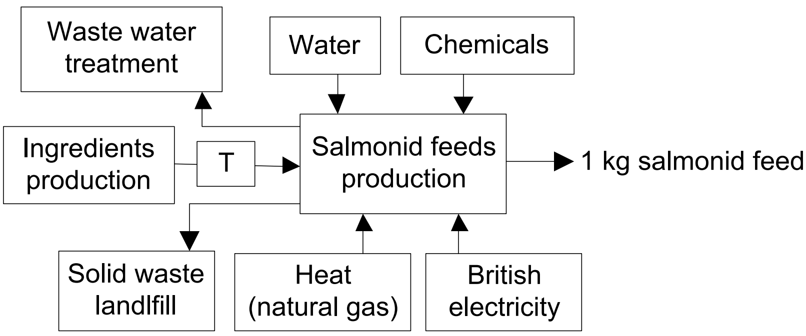


Figure 39 Salmonid feed product system (T: transport)

The functional unit of the system was 1 kg of each feed formulation at the plant gate. The calculation was assisted by the Simapro 7.3 software package (PRe Consultants, 2011). Life cycle GHG emissions results were obtained using the Greenhouse Gas Protocol 1.01 impact assessment method (The Greenhouse Gas Protocol, 2010). Biogenic CO_2

emissions (e.g. from the combustion of rendered products - Chapter 4) were not included as it was considered they do not result in a net gain of CO_2 in the atmosphere.

Table 34 Salmonid feed formulations used in the study

Ingredients	Conventional	Maximised poultry processed animal protein (Max PPAP)	Maximised hydrolysed feather meal+ porcine haemoglobin meal (Max HFM+PHM)	Maximised fish meal (Max FM)
Fish meal	20.00	15.00	15.00	45.45
Poultry processed animal protein		16.00		
Hydrolysed feather meal			10.00	
Porcine haemoglobin meal			4.00	
Vegetable ingredients	47.95	41.25	41.3	29.17
Fish oil	29.77	27.27	28.65	25.05
Other components[a]	2.28	0.48	1.05	0.33

[a] Other components: minerals and vitamins

GHG emissions from poultry by-product derived ingredients (poultry by-product meal and hydrolysed feathers meal) were taken from Chapter 7. It is recognised that in the UK, the GHG emissions associated with poultry by-product derived ingredients depend on the type of fuel used in their processing (category 1 mammalian rendered fat (MRF) or natural gas); therefore formulations that included these ingredients were evaluated when either 100% of thermal energy used for ABP rendering was derived from MRF or natural gas.

The product system used in this study for the production of porcine haemoglobin meal is presented in Figure 40. GHG emissions associated with UK pig production were taken from Williams *et al.* (2006). The model was developed further under the DEFRA-funded project IS0222 and modified for this study to express environmental burdens on a live-weight rather than deadweight basis as 3.027 kgCO_2e per kg live weight pig at the farm gate (AG Williams, pers. comm., 2011). Data on energy use in pig slaughtering were taken from The Finnish Environment (2002). Data on the yields of different co-products and waste flows from pig slaughtering were taken from EBLEX (2006); a summary of these product flows is presented in Table 35. Economic flows in porcine blood processing were provided by one UK porcine blood processing plant (Table 36).

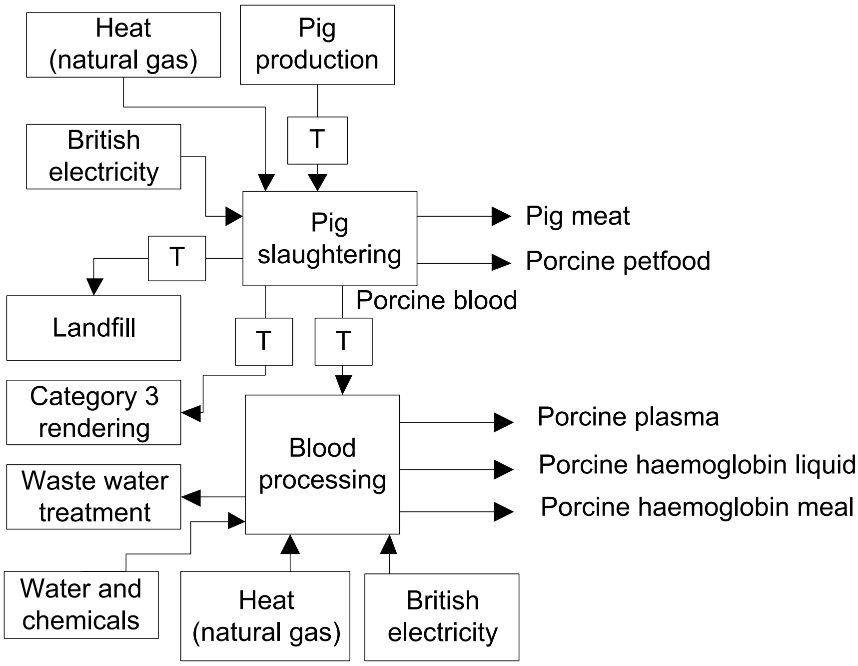


Figure 40 Porcine processed by-products system (T: transport)

Table 35 Economic flows for the slaughtering of 1 kg of live weight pigs (The Finnish Environment, 2002; EBLEX, 2006)

Co-products	Units	
Pig meat	kg/kg	0.699
Porcine petfood	kg/kg	0.014
Porcine blood	kg/kg	0.041
Inputs/Outputs		
Electricity	kJ/kg	972
Heat (as content in natural gas)	kJ/kg	1,728
Category 3 porcine by-product	kg/kg	0.147
Waste to landfill	kg/kg	0.1

Table 36 Economic flows for the processing of 1 kg of porcine blood for one UK blood processing plant

Co-products	Unit	
Porcine Plasma	kg/kg	0.049
Porcine Haemoglobin Meal	kg/kg	0.123
Porcine Haemoglobin Liquid	kg/kg	0.042
Inputs/Outputs		
Electricity	kJ/kg	267.4
Heat (as content in natural gas)	kJ/kg	766.8
Water	kg/kg	1.274
Chemicals	kg/kg	0.000696
Waste water	m3/kg	0.001274

Sources of fish biomass to produce fish meal and fish oil may be very different (Pelletier *et al.*, 2009; Peron *et al.*, 2010). Therefore two main sources of fish derived ingredients were evaluated for each formulation: fish meal and oil derived from fish by-products (fish trimmings from the white fish processing industry for human consumption) and derived from small pelagic species fisheries. The latter are the most common source of fish biomass for the production of fish meal and oil (Peron *et al.*, 2010). The life cycle inventory for the production of fish meal and oil derived from a small pelagic species (sand eel) was taken from Nielsen *et al.* (2003), but modified to use British electricity and to express the results based on allocation. In the original system modelled, system expansion was used to include the avoidance of rapeseed oil by the co-production of fish oil. This modification was required as both fish meal and oil are required in the salmonid feed product system modelled.

Figure 41 illustrates the fish meal and oil product system based on white fish by-product. The life cycle inventories for white fish fishing (assumed Danish cod) and processing were taken from Danish food LCA databases (Nielsen *et al.*, 2003). Data on energy use and production yields of fish meal and oil during the rendering of fish by-products were provided by a UK plant (Table 37).

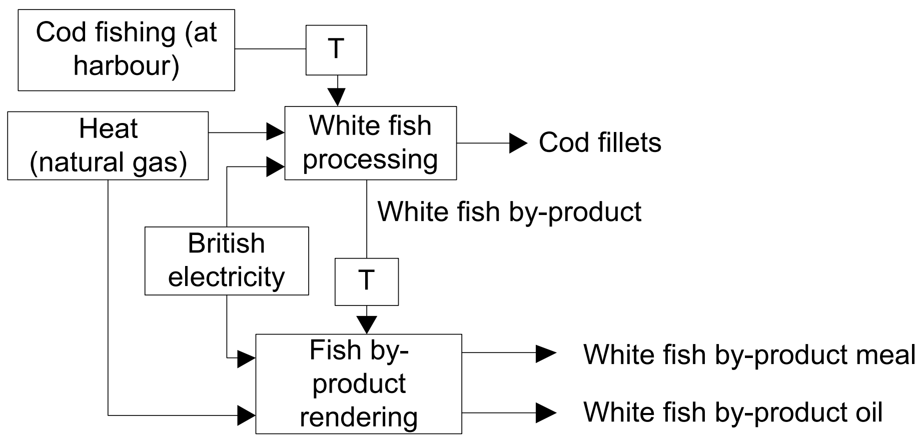


Figure 41 White fish by-product meal and oil product system (T: transport)

Table 37 Economic flows for the rendering of 1 kg of fish by-product for one UK by-product processing plant

Co-products	Unit	
Fish by-product meal	kg/kg	0.205
Fish by-product oil	kg/kg	0.09
Inputs/Outputs		
Electricity	kJ/kg	209
Heat (as content in natural gas)	kJ/kg	1,512

Life cycle inventories for the production of wheat, beans, corn, sunflower seeds, and Dark Distilled Grains (DDGS) (a by-product of ethanol) from rye were taken from Ecoinvent databases (Ecoinvent Centre, 2010). Sunflower seed processing co-product yields (oil and meal) and electricity and heat inputs in processing were taken from Ragaglini *et al.* (2011). Energy (electricity and heat) use associated with the processing of corn to co-products (starch, oil, gluten feed and gluten meal) was taken from Pelletier (2006). The corn process co-products and by-products yields were taken from ERS/USDA (2011).

The soya bean to soya meal and oil processing life cycle inventory was taken from Nielsen *et al.* (2003) and modified to use Brazilian soya beans from Ecoinvent databases (Ecoinvent Centre, 2010). For the latter, GHG emissions associated with land transformation were also included. Data on energy use and yield of soya protein concentrate during the processing of soya bean meal was assumed to be similar to the processing of peas to peas protein concentrate and was taken from Apaiah *et al.* (2006), as performed by Pelletier *et al.* (2009).

Life cycle inventories for British electricity, heat from natural gas, chemicals, water, waste water treatment, landfill and transport were taken from Ecoinvent databases (Ecoinvent Centre, 2010). Road transport distance in every instance required was assumed as 200 km.

8.4.2 Co-product handling combinations

A co-product can be defined as "any of two or more products coming from the same unit process or product system" (ISO, 2006a). A by-product is "a secondary product obtained during the manufacture of a principal commodity" (Meeker and Hamilton, 2006). Three co-product handling combinations were employed in the current study:

- Mass allocation (MA): All the co-product handling problems in the system were solved based on mass of co-products and by-products.
- Economic allocation (EC): All the co-product handling problems in the system were solved based on the mass weighted economic value of the co-products and by-products.
- Economic allocation – mass allocation (EC-MA): The co-product handling for the separation of co-products from by-products (fish meat from fish by-products, pig meat from pig by-products, poultry meat from poultry by-products, sunflower oil from sunflower meal, corn starch from corn by-products, ethanol from DDGS, soya bean meal from soya oil, wheat grains from wheat straw) was performed based on the mass weighted economic value of co-products and by-products. Further co-product handling (if needed) was performed based on mass allocation (fish by-product rendering co-products, porcine blood processing co-products, poultry by-products processing co-products). Co-product handling in systems only processing co-products (small pelagic species rendering co-products) was based on economic allocation; under this co-product handling combination it is arguable that this could have been performed based on mass; however for these co-products both factor sets are similar.

There is one exception to these co-products handling combination rules. The life-cycle inventory for cod landed at the harbour reported by Nielsen *et al.* (2003) includes system expansion to account for avoidance of the by-catch.

Table 38 presents the allocation factors used in the study. Allocation factors associated with poultry by-product derived ingredients were presented in Chapter 7. The only ingredient not requiring co-product handling was beans. Ecoinvent databases are based on economic allocation. When required for wheat, DDGS, sunflower meal, the allocation was modified to be based on mass.

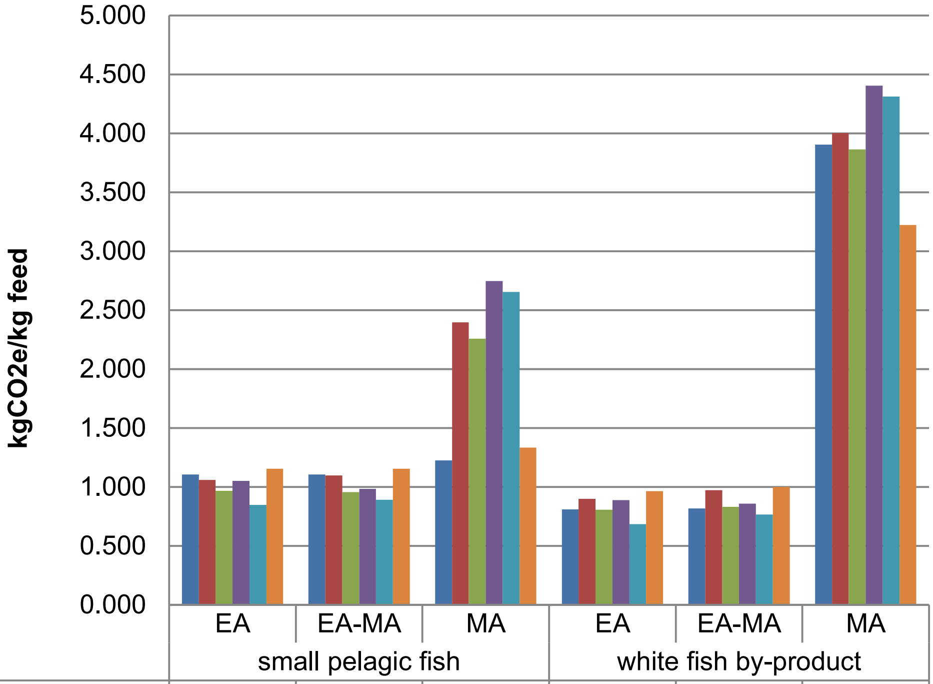
Table 38 Allocation factors used in the product system

Process	Co-products and by-products	Allocation factors		Data Sources
		Mass	Economic	
White fish by-product rendering	Fish meal (white fish by-product)	69.49	76.98	Mass: yields Price: Fish meal and soya bean oil (assumed as fish oil) (Indexmundy, 2011)
	Fish oil (white fish by-product)	30.51	23.02	
White fish processing	White fish fillet	50.00	98.20	Mass and prices: personal contact with UK white fish processing companies
	White fish by-product	50.00	1.80	
Sand eel rendering	Fish meal (small pelagic species)	82.78	87.65	Mass: yields Price: Fish meal and soya bean oil (assumed as fish oil) (Indexmundy 2011)
	Fish oil (small pelagic species)	17.22	12.35	
Porcine plasma processing	Porcine plasma	23.00	67.57	Yields and prices provided by blood processer
	Porcine haemoglobin liquid	19.49	1.23	
	Porcine haemoglobin meal	57.52	31.20	
Pig slaughtering	Pig meat	92.82	99.83	Yields: (EBLEX, 2006) Prices: (EBLEX, 2006; BPEX, 2011)
	Porcine pet food	1.80	0.12	
	Porcine blood	5.39	0.05	
Soya bean milling	Soya bean meal	82.14	63.37	Yields Ecoinvent(Ecoinvent Centre, 2010) (Indexmundy, 2011)
	Soya bean oil	17.86	36.63	
Corn processing	Corn starch	64.02	59.81	(ERS/USDA, 2011)
	Corn oil	3.15	14.88	
	Corn gluten feed	27.44	14.68	
	Corn gluten meal	5.39	10.64	
Sunflower seed processing	Sunflower oil	35.60	76.18	Yields (Ragaglini *et al.*, 2011) Prices: (Agri Commodities, 2011) (Forex, 2011)
	Sunflower meal	64.40	23.82	
Processing of rye grains	Ethanol	50.00	97.70	Ecoinvent(Ecoinvent Centre, 2010)
	DDGS[a]	50.00	2.30	
Wheat production	Wheat grains	62.14	92.50	Ecoinvent(Ecoinvent Centre, 2010)
	Wheat straw	37.86	7.50	

[a]DDGS: Dark Distilled Grains

8.5 Results

Figure 42 presents the GHG emissions for each salmonid feed formulation based on both small pelagic species and white fish by-product derived ingredients and for the three co-product handling approaches.



	small pelagic fish EA	small pelagic fish EA-MA	small pelagic fish MA	white fish by-product EA	white fish by-product EA-MA	white fish by-product MA
Max FM	1.105	1.105	1.226	0.809	0.817	3.904
Max PPAP (f:NG)	1.058	1.096	2.397	0.900	0.972	4.002
Max PPAP (f:MRF)	0.966	0.956	2.258	0.807	0.832	3.863
Max HFM (f:NG) + PHM	1.050	0.983	2.746	0.887	0.858	4.404
Max HFM (f:MRF) + PHM	0.848	0.891	2.655	0.685	0.765	4.312
Conventional	1.155	1.155	1.333	0.965	1.000	3.223

Figure 42 Greenhouse gas emissions associated with the production of salmonid feed (kgCO_2e/kg feed) (EA: economic allocation, EA-MA: economic allocation – mass allocation, MA: mass allocation, Max FM: Maximised fish meal, Max PPAP (f: NG): Maximised poultry processed animal protein produced using natural gas as fuel in poultry by-product processing, Max PPAP (f: MRF): Maximised poultry processed animal protein produced using mammalian rendered fat as fuel in poultry by-product processing, Max HFM (f:NG) + PHM: Maximised hydrolysed feather meal produced using natural gas as fuel in poultry by-product processing + porcine haemoglobin meal, Max HFM (f:NG) + PHM: Maximised hydrolysed feather meal produced using mammalian rendered fat as fuel in poultry by-product processing + porcine haemoglobin meal)

Results based on small pelagic species fish biomass ranged between 0.848 and 1.155, 0.891 and 1.155, and 1.226 and 2.746 kgCO_2e/kg feed for EA, EA-MA, and MA co-product handling combinations respectively. When EA or EA-MA were employed the minimum GHG intensity was calculated for the formulation Max HFM+PHM based on MRF as a fuel for ABP processing and the maximum GHG emissions intensity was calculated for the Conventional formulation. However, when MA was employed the minimum GHG intensity was calculated for the formulation maximised FM and the maximum GHG intensity was calculated for the Max HFM+PHM based on MRF as a fuel for ABP processing. For Max PPAP, GHG emissions where 9%, 13% and 6% lower when MRF was used as a fuel in ABP processing in comparison to natural gas for EA, EA-MA, and MA co-product handling combinations respectively. For Max HFM+PHM, GHG emissions where 19%, 9% and 3% lower when MRF was used as a fuel for ABP processing in comparison to natural gas for EA, EA-MA, and MA co-product handling combinations respectively.

Results based on white fish by-product biomass ranged between 0.685 and 0.965, 0.765 and 1.000, and 3.223 and 4.404 kgCO_2e/kg feed for EA, EA-MA, and MA co-product handling combinations respectively. When EA or EA-MA were employed the minimum GHG intensity was calculated for the formulation Max HFM+PHM based on MRF as a fuel for ABP processing, and the maximum GHG intensity, was calculated for the Conventional formulation. However, when MA was employed the minimum GHG intensity was calculated for the Conventional formulation and the maximum GHG emissions intensity for the Max HFM+PHM based on natural gas as a fuel for ABP processing. For Max PPAP, GHG emissions where 10%, 14% and 3% lower when MRF was used as a fuel in ABP processing in comparison to natural gas for EA, EA-MA, and MA co-product handling combinations respectively. For Max HFM+PHM, GHG emissions where 23%, 11% and 2% lower when MRF was used as a fuel for ABP processing in comparison to natural gas for EA, EA-MA, and MA co-product handling combinations respectively.

When employing EA and EA-MA, each feed including fish biomass derived from white fish by-products had lower GHG emission intensity in comparison to those with fish derived biomass from small pelagic species (17% on average). In contrast, when employing MA, each formulation including fish biomass derived from small pelagic species had a lower GHG emissions associated in comparison to fish derived biomass from white fish by-products (48% on average).

8.6 Discussion

8.6.1 Formulations

All the formulations evaluated in this study (Table 34) used high inclusion rates of fish derived ingredients (between 25.05 % and 29.77 %, and 15.00 % and 45.45 % fish oil and meal respectively). The maximised fish meal formulation had a total inclusion rate of fish derived ingredients of 70.50 %. The remaining three formulations had fish biomass derived ingredient inclusion rates of between 42.27 % and 49.77 %. Formulations with maximised inclusion of terrestrial animal by-product derived ingredients had the lowest inclusion rates of fish derived ingredients. The conventional formulation had the highest inclusion rate (47.95 %) of vegetable derived ingredients and 49.77 % of fish derived ingredients.

Pelletier *et al.* (2009) used a total inclusion rate of vegetable derived ingredients of 33.5 % and fish derived ingredients of 66.5 % for the average UK salmonid feed formulation in 2007. The formulation utilised by Pelletier *et al.* (2009) had a higher inclusion rate of fish derived ingredients than the conventional in the current study. Tacon and Metian (2008) estimated inclusion rate ranges of fish meal and fish oil of between 25 % and 46 % and 20 % and 35 %, respectively (total fish biomass derived ingredients of between 45 and 64%) in UK salmon feed production in 2006. Both Pelletier *et al.* (2009) and Tacon and Metian (2008) agreed that of the countries studied UK farmed salmon are fed with the highest proportion of fish derived biomass.

The feeds used in the current study were formulated to evaluate the inclusion of two contrasting sources of fish derived ingredients and terrestrial animal by-product derived ingredients and therefore can be considered hypothetical. The fish derived ingredient inclusion rates are in concordance with the ranges for UK salmonid feeds reported by Tacon and Metian (2008).

8.6.2 Greenhouse gas emissions

The GHG emissions for the formulations investigated ranged between 0.685 and 4.404 kgCO_2e per kg salmonid feed (Figure 42), which are in the same order of magnitude as ranges in the literature. Pelletier and Tyedmers (2007) calculated GHG emissions between 0.690 and 1.400 kgCO_2e per kg feed, Pelletier *et al.* (2009) between 1.430 and 2.290 kgCO_2e per kg feed, Papatryphon *et al.* (2004) between 1.120 and 1.560 kgCO_2e per kg feed, and Boissy *et al.* (2011) between 1.45 and 1.96 kgCO_2e per kg feed.

The highest GHG emissions were obtained for each formulation based on mass allocation and white fish by-product derived ingredients, principally because white fish fishing is highly energy intensive and the use of mass allocation partitions the GHG emissions equally between all the outcomes of fish processing (co-products and by-products). With this co-product handling approach and source of fish biomass, the conventional formulation had the lowest GHG associated emissions, mainly because this formulation had the highest inclusion rate of vegetable derived ingredients.

There is a considerable difference in the results when using mass allocation regarding the source of fish biomass. The main reason for this is energy use in fishing. White fish fisheries use significantly more energy than small pelagic species fisheries. Using the life cycle inventories for the fishing of cod and sand eel (Nielsen *et al.*, 2003), the GHG emissions associated with 1 kg of landed fish were calculated as 1.18 and 0.17 kgCO_2e respectively. In addition, the energy used in the rendering of sand eel is 1,332 and 145 kJ thermal energy and electricity respectively per kg (Nielsen *et al.*, 2003). These values are lower than those reported here for rendering of fish by-products in the UK (1,512 and 209 kJ thermal energy and electricity respectively per kg (Table 37)). In addition, the product system for fish meal and oil derived from fish by-products included the fish processing stage, where the fish fillets (edible parts) are separated from the fish by-products (Figure 41). Small pelagic species are fished for the production of fish meal and oil and therefore are sent directly from the harbour to the fish rendering plant.

When mass allocation was employed formulations that included porcine haemoglobin meal had the highest GHG emissions, regardless of whether small pelagic species or white fish by-products were used as the source of fish biomass. With this co-product handling approach porcine haemoglobin meal was the ingredient with the highest GHG associated emissions (20.307 kgCO_2e/kg). This occurred because the GHG emissions associated with pig production and slaughtering are distributed among all the co-products and by-products based on mass. In addition, since blood has a relatively low dry matter, the yield of blood derived products (plasma, haemoglobin liquid and haemoglobin meal) is 0.21 kg products per kg of blood (Table 36). In contrast, GHG emissions of 0.691, 0.375 kgCO_2e/kg porcine haemoglobin meal were calculated when employing EA-MA and EA, respectively because when using economic allocation the contribution of pig production and slaughtering was very low (Table 38).

When the source of fish ingredients was small pelagic species and mass allocation was used, both conventional and maximised fish meal had similar and relatively low GHG emissions. With mass allocation terrestrial animal by-product derived ingredients had relatively high GHG emissions. An insight on the GHG of poultry by-product derived ingredients with different co-product handling approaches can be found in Chapter 7.

The results for economic allocation and economic-mass allocation were very similar; this was expected because by-product derived co-products had similar prices on a mass basis. Contrary to mass allocation, GHG emissions based on these approaches were lower when the source of fish derived ingredients was white fish by-products in comparison to small pelagic species. When economic allocation was used, formulations that included terrestrial derived animal by-products had lower GHG emissions than the conventional formulation. With this approach, the conventional formulation had the highest GHG emissions for each source of fish derived ingredients. The use low value by-product biomass results in Climate Change benefits when employing economic allocation (Table 38).

For each formulation and co-product handling approach that included poultry by-product derived ingredients, the GHG emissions were lower when these ingredients were produced using rendered fats instead of natural gas as a fuel in the rendering process (Chapter 7).

All the formulations included at least 42.27 % and up to 70.50 % ingredients derived from fish biomass, and therefore the GHG emissions associated with these ingredients were very important in determining the GHG intensity of each formulation. When co-product handling was based on mass, using a low energy dedicated small pelagic fishery derived product instead of a high energy fishery derived white fish by-product as source for fish derived ingredients resulted in lower GHG emissions. This is in agreement with the findings of Pelletier and Tyedmers (2007) using gross energy based allocation. Similarly, when co-product handling was based on economic value, using a low value fish by-product derived product instead of small pelagic species as source for fish derived ingredients resulted in lower GHG emissions. This is in agreement with the findings of Papatryphon *et al.* (2004) using allocation based on economic value.

It should be noticed that there are three rendering processes included in the different feed production systems: the poultry by-product, the white fish by-product and the small pelagic species rendering. The results when using economic allocation are very similar for formulations including fish by-product and poultry by-product derived ingredients. Variation in energy intensity in rendering can be important, for example, rendering of mammalian material can have different energy intensity (Chapter 4). Therefore it is difficult to provide a strong conclusion about whether it is better from a Climate Change perspective to use either fish by-product or terrestrial animal by-products in farmed salmon feeds when employing economic allocation (as the contribution from production and processing is very low). However it is clear that with this co-product handling approach, ingredients derived from animal by-products (terrestrial or aquatic) have a relatively low GHG emission intensity in comparison to other ingredients in the system (e.g. small pelagic species).

The focus of this study has been the replacement of protein sources, while the fish oil inclusion rate has been kept similar in each feed. Fish oil and meal are co-produced from the same material. Use of fish oil without concurrent use of fishmeal would not seem ideal. The GHG emissions of replacing fish oil with vegetable derived oils has been studied by Boissy *et al.* (2011).

The "Other components" (vitamins and minerals) have not been included in the assessment of GHG emissions of the different formulations. The Conventional formulation has the highest inclusion rate of these components (2.28%) (Table 34). It is not expected that their inclusion would radically affect the GHG emission comparison performed in this study.

8.6.3 Co-product handling and the use of animal derived by-products in feeding farmed animals

The use of terrestrial animal derived by-products in feeding farmed animals (water or land based) is currently not permitted in the EU (EC, 2009). However, the European Commission is currently re-considering the use of poultry derived processed animal protein to farmed pig and farmed fish and pig derived processed animal protein to farmed birds and farmed fish (fish derived material is already allowed to be use in the feeding of non-ruminants) (Gleadle, 2011). When economic allocation was used to partition the inputs and outputs in the product system, the inclusion of poultry by-product, porcine blood, and fish by-product derived ingredients in salmonid feed formulations resulted in lower GHG emissions than conventional formulations. When mass allocation was used the situation was different and salmonid feeds derived from vegetable protein sources and low energy intensive dedicated reduction fisheries had lower GHG intensity.

The results for mass allocation are in agreement with the main findings by Pelletier and Tyedmers (2007) and Pelletier *et al.* (2009) based on gross energy content allocation. Mass allocation factors are in very good agreement with energy allocation factors, as presented by Pelletier and Tyedmers (2007), which is reasonable as it can be expected that for livestock systems the chemical composition of the edible part will be fairly similar to that of the animal by-products resulting in similar allocation factors for energy or mass based allocation. Ayer *et al.* (2007) and Pelletier and Tyedmers (2007) argued that economic allocation is not ideal, as economic values reflect market failures and do not reflect the material and energy flows in co-product streams. However, Ayer *et al.* (2007) also acknowledged that from a conservation perspective the use of economic allocation may seem appealing, as the economic revenue is the driver for production. This argument would seem appropriate in the case of animal by-products. In general, the basis for assessment based on products is that the environmental impact of the economy is related to the consumption of products (and services), directly by their use and indirectly by their production and final disposal (Tukker and Jansen, 2006).

Based on LCA results some studies have recommended that salmonid diets should be more vegetarian (Ellingsen and Aanondsen, 2006; Pelletier and Tyedmers, 2007). Although this seems in principle appealing, demand for animal products is increasing and the production of animal by-products is unavoidable. Animal by-product prices may vary over time; however it is not likely that their demand will influence their volume of production. Their production volume is dominated by the demand of the main commodity (the edible animal parts). With an increasing demand for animal products and provided that animal by-products are not being used for human consumption, it seems logical that they are used to minimise the need (at least partially) for marginal proteins and fats such as soya bean meal and palm oil, which incur GHG emissions associated with land transformation.

Global meat production increased from 233 to 279 million tonnes between 2000 and 2008 production (FAO, 2011); an increase of 19%. This trend is expected to continue with meat production projected to more than double from 1999-2001 to 2050 (Steinfeld *et al.*, 2006). This will not only result in a significant increase in the production of animal by-products, but will also likely result in more forest land being converted to pastures and arable land to produce feed ingredients. This land transformation could be minimised through more widespread use of animal by-products. For example, animal by-products could replace at least partially the use of marginal soya bean meal in the UK salmonid farming sector. However, it is likely that the soya bean meal would be used in another livestock production system or in the petfoods sector (where animal by-products are currently mostly used). Consequently the major problem is not the production of marginal feed ingredients itself, but the global growth of the animal production system. This provides a defensible argument to use economic allocation to divide the environmental burden of animal products and by-products, as the driver for its expansion is the delivery of animal food for human consumption.

From a research perspective, it seems that each co-product handling situation should be analysed independently, however it is has been argued that there is a need to harmonise co-product handling (Flysjö *et al.*, 2011a). The British Standard for *Carbon Footprint* of products and services (BSI, 2008b) prescribes that when allocation cannot be avoided, it should be based on economic value, indicating that in attributional *Carbon Footprint* studies in the UK, it is likely that economic allocation would be the method of choice. Recently Boissy *et al.* (2011) evaluated the partial replacement of small pelagic species derived fish biomass with vegetable derived material in the feeding of farmed salmonids using economic allocation as the main method for co-product handling (they did not include either aquatic or land based animal by-products). Economic allocation has been also used as co-product handling method in a recent study on poultry feeds production (Nguyen *et al.*, 2012).

It has been shown that co-product handling is in fact a very important issue when analysing the life cycle impact assessment results of the inclusion of animal by-products derived ingredients in farmed animals feed formulations (in particular salmonid feeds in the current study). However specific practices of some parts of the production chain should be considered. For example, the processing of animal by-products uses significant proportions of rendered fats (biofuels) instead of fossil fuels to fulfil their thermal requirements in the UK (Chapter 4). This is an issue that should be taken into account when quantifying the Climate Change impact from animal by-product derived ingredients.

It is acknowledged that there are no technical impediments for the use of biofuels in the production chain of other ingredients (e.g. in the rendering of fish biomass), however this is a mature practice associated with the terrestrial animal by-product rendering sector, as low value rendered fats are actually products of the terrestrial animal by-products processing industry. It is important to notice that the use of rendered fats may vary from plant to plant and year to year, therefore scenarios with different fuels are useful to have a broad perspective on this issue.

8.6.4 Additional considerations

In the current study the only environment impact category studied was Climate Change. Other environmental impacts should be considered when the environmental and resource sustainability of farmed (aquatic or terrestrial) animals (or feeding them) is studied. The availability of animal by-product biomass to replace significant amounts of small pelagic species has not been evaluated. Furthermore, the shift of animal by-products from being used as ingredients in petfoods to animal feeds would require consequential assessment to quantify net environmental benefits. A broader perspective in sustainability assessment would include not only the environmental but also the social and the economic dimensions (Guinee *et al.*, 2010). The use of caught fish in feeding farmed fish is perceived as ecologically inefficient and seems associated with an social issue when there is competition for small pelagic species for either food for vulnerable parts of the society in developing countries or feed for high value fish species (Tacon and Metian, 2009; Bostock *et al.*, 2010).

8.7 Conclusion

The inclusion of animal by-product derived ingredients in salmonid feed formulations results in lower GHG emissions on a feed mass produced basis when they substitute part of vegetable or small pelagic species fish derived ingredients. These findings are strongly dependent on the co-product handling method used to partition between edible animal products and by-products (economic allocation). GHG emissions of feed formulations including land-based animal by-products are reduced when rendered fats are used as fuel for rendering instead of fossil fuels because of the emissions credits associated with the rendering of null value animal by-products. Economic allocation is perceived as an appropriate co-product handling approach for this type of system since the need for marginal feed ingredients is driven by the global expansion of the livestock production sector. Consequential assessment would be required to prove net benefits of a change in the destiny of ABP derived products.

Chapter 9

General discussion

9 General discussion

9.1 Novel contribution

In general rendering is not a commonly known industry (publicly and academically). Prior to this research study the energy use and GHG emission intensity of the UK rendering process (presented in Chapters 4 and 7) had not been investigated and thus the research presented can be considered a novel contribution to knowledge. Rendering is the main process by which a significant amount of biomass arising from the animal production and meat processing industries (30 – 50% the live weight produced) is processed. It is therefore important to quantify its Climate Change relevance in the context of animal production (presented in Chapters 5 and 7 – system expansion).

In addition, the research presented in this thesis is considered an important contribution to knowledge because it provides a robust source of data on economic flows for the rendering process. These economic flows can be used in life cycle assessment studies of rendered products (for the production of pet foods, animal feeds and biofuels) and can be included in system expansion in LCAs of animal products. Consequently in addition to the direct contribution to knowledge contained in this thesis, the research has the potential to contribute to knowledge indirectly (i.e. through the work of others). For example, the un-allocated economic flows in rendering reported in Chapter 4 can be used in LCA studies on biodiesel derived from rendered fats using similar or different co-product handling approaches and system boundaries to those used here.

Broiler production is one of the most important meat production systems (representing 40% of the meat produced in the UK); however studies on its GHG emissions are not abundant. Furthermore, there have been no published LCA studies on the GHG emissions of broiler production based on direct farm data in the UK. Chapter 6 constitutes one study of the fossil energy use and GHG emission intensity of conventional broiler production in the UK. Chapter 6 adds to the publicly available literature on economic flows, energy use and Climate Change impact of meat production systems.

Co-product handling (e.g. allocation and system expansion) is one of the most discussed topics in LCA methodology. Chapter 7 presented a new case study on different ways to deal with the co-production problem in poultry processing systems and thus demonstrates an important contribution to knowledge. There are no previous peer reviewed studies which have included a detailed discussion of co-product handling in the poultry production system. In fact, there are no detailed studies on poultry products and by-products that present simultaneous LCA results for final poultry meat products and rendered products.

Published LCA studies on the production of salmonid feeds, and in particular the inclusion of animal by-products as ingredients, had not previously taken into account the fact that rendered products can be produced with rendered fats as fuels for the rendering process. Chapter 8 presented that in spite of the previously discussed issues in the literature related to co-product handling, it is also important to take into consideration the use of rendered fats as fuels for the rendering processes. The use of rendered fats as fuels for rendering resulted in GHG emission credits.

Although it was not an objective of the thesis per-se, the potential for energy self sufficiency of the UK rendering industry was screened. The procedure for estimation can be found in Appendix C. The UK rendering industry produces different categories of rendered products. Under current EU legislation category 1 rendered products must be destroyed by combustion. During the years 2006 to 2008 sufficient quantities of category 1 rendered products were produced to satisfy both the thermal energy and electricity requirements of the UK rendering industry. This means that the rendering industry is not necessarily an energy consumer, but an energy producing system and this is the main reason why they provide GHG emission credits (provided rendered products substitute products or energy systems with higher GHG emission intensity).

9.2 Additional considerations

9.2.1 Co-product handling and the animal by-product system

One important methodological choice was the use of economic allocation and the consequential system boundary establishment for mammalian and poultry rendering systems (Section 2.3.4). It should be noted that if mammalian ABP had a positive price, mammalian rendered products would need to be modelled using a similar approach to poultry derived rendered products (Chapter 7). In addition the treatment of MBM has been included in the category 1 rendering product system because it was not associated with a positive economic value. If this situation were to change, the system would not gain credits from the avoidance of highly carbon intensive electricity. The effect on category 3 mammalian rendered products would be similar to that of using carbon neutral electricity as avoided electricity presented in the sensitivity analysis of Chapter 4 (GHG emissions of category 3 mammalian rendered products would be higher).

If a new EU risk assessment permitted the inclusion of poultry rendered products in non-ruminant farm animal diets (Gleadle, 2011), it may have an effect on their prices. This would possibly require an update in the prices and consequently the economic allocation factors in Chapter 7 would need to be revised. However, the economic allocation factors would not change radically as the price of the edible part is (and is likely always to be) significantly higher than that of the by-products.

Increase in biodiesel production could create a demand for rendered fats to be used in the production of biodiesel. If all the rendered fats produced were to be used in the production of biodiesel, the rendering industry would have to rely on fossil fuels for their fuel requirements, with the consequential effect on the GHG emissions from rendered products (Chapters 4 and 7) and meat production systems (Chapters 5 and 7).

When using system expansion, soya bean has been used as a marginal source of protein and oil, or palm oil as the source of oil. It has been assumed that the use of rendered products would be associated with the avoidance of the production of these marginal crop derived ingredients. Both soya bean and palm have relatively high GHG emissions because their production is associated with agricultural emissions and with land transformation. In the case of soya bean meal as a source of protein, it is difficult to assure a net avoidance of soya bean meal production. If soya bean meal is partially displaced by poultry animal processed protein and hydrolysed feather meal, another source of protein would be needed in the production of petfoods (where the poultry meals are currently being used). It is likely that petfood manufacturing would use a marginal protein (possibly soya bean meal). Therefore careful examination of the consequential effects on different systems is needed to prove net GHG emission reductions.

The inclusion of ABP management in the meat production system provided GHG emission credits to the edible products (Chapters 5 and 7 – system expansion). However, the effect of the fuel used in rendering was different for beef (Chapter 5) and chicken (Chapter 7), because in the former the rendered fat was produced within the same system, whist in the latter the rendered fat was outsourced (from the category 1 rendering system – Chapter 4).

A question that arises: What is the ideal final use of animal by-products from a Climate Change perspective? To answer this question would require the whole system (including the use of animal by-products and substitutes) to be modelled because effects in one system can have knock on effects on others. The integrated modelling of the GHG emissions of the whole food-petfood-oleochemicals system would minimise the need for co-product handling. This approach was neither in the objectives nor scope of this research project. The need for an integrated systems analysis has already been addressed for important interlinked protein systems that include milk and beef (Flysjö *et al.*, 2011a).

In the current work economic allocation has been perceived as the most adequate co-product handling method when results for GHG emissions associated with rendered products are required, for example when the GHG emissions of an application is being calculated (e.g. animal diets - Chapter 8). However, it is important to note that production of animal by-products is an important part of the food chain and therefore it could be argued that the most appropriate way of dealing with the animal by-product processing system is to include it as part of the meat production system as in Chapter 5 and Chapter 7 (system expansion). This is in fact closer to the ideal case of modelling the whole food production system. Furthermore, system expansion is in a higher tier than allocation as recommended by the ISO standards for LCA (ISO, 2006b; a) and de BSI Standards for *Carbon Footprint* (BSI, 2008b).

9.2.2 Other alternatives to treat animal by-products

The research presented did not attempt to compare the Climate Change impact (in terms of GHG emissions per kg ABP) of treating animal by-products by rendering against alternative disposal methods (e.g. centralised and decentralised anaerobic digestion, the biomal process, direct incineration). This would be particularly important for the evaluation of alternative disposal options for non-ruminant on-farm fallen stock. Non-ruminant mortalities (Category 2 animal by-products) are permitted to be used as feedstock for anaerobic digestion. Anaerobic digestion or co-digestion appear to be an appealing option to transform solid organic waste into biogas because they can use a great variety of feedstock materials (Khalid *et al.*, 2011). The biomal process consist in fine crushing the animal by-products and co-combusting them together with other fuels (e.g. wood chips, peat or municipal waste) in a fluidised bed combustion system to produce both heat and electricity (Biomal).

In contrast to Category 2 material, legislation requires Category 1 animal by-products to be incinerated directly or after rendering. Whilst the current study has investigated the Climate Change impact of using Category 1 rendered products as fuels (as presented in Chapter 4), it has not attempted to compare the impact of generating energy through the direct incineration of Category 1 slaughterhouse material (i.e. material that has not been rendered).

Currently category 3 rendered products are used in the oleochemical industry and petfoods, but could also be used in farmed animal feeds provided that new risk assessments indicate that this is acceptable from an animal and human health perspective. In particular, that their use as feed ingredients does not constitute a risk of disease transmission. Rendering is the main technology used to dry animal by-products to produce feed ingredients. In fact, rendering technology is even used in the production of fish meal and oil from dedicated small pelagic reduction fisheries. Consequently, it seems that there are no other alternative treatment methods for Category 3 material (that is to be used as feed ingredients) whose Climate Change impact could be compared to that of the rendering process. Improvements within rendering technology have not been studied.

9.2.3 Other environmental impact categories

Only the Climate Change impact has been assessed in this study. Climate Change is currently a priority in environmental policy, however it can be argued that environmental sustainability is not only about GHG emissions. In this study, one important issue that has driven the conclusions has been the use of rendered fats as fuel for the production of rendered products. The use of rendered fats as fuels has only been assessed with a Climate Change perspective.

Previous research has indicated that use of biofuels may either increase or decrease NO_x emissions. The use of biodiesel derived from vegetable fats as fuel for internal combustion engines causes reduced tailpipe emissions of all pollutants, with the exception of NO_x that has been shown to increase depending on the level of substitution (Hansen *et al.*, 2006; Mirheidari *et al.*, 2012). In contrast the use of biofuels derived from crops and crop residues in boilers results in reduced NO_x emissions (Saidur *et al.*, 2011).

Therefore research is needed to comprehensively characterise the emissions produced from the combustion of rendered fats and MBM in order to fully understand their environmental impact. NO_x emissions are associated with several impact categories that include: Acidification, Eutrophication, and Photo-oxidant formation. The current work has not endeavoured to evaluate these indicators.

9.2.4 Recommendations for further work

The recommendations for further work related to this thesis are presented below:

- The inclusion of other impact categories. As discussed in section 9.2.3 a more comprehensive approach would include more impact categories, in particular those associated with emissions from the combustion of biofuels. The economic flows in the rendering process have already been presented in this work. Therefore this would require the inclusion of natural flows associated with additional impact categories. In the particular case of NO_x, data on emissions from the combustion of rendered products in industrial boilers and FBC systems would be needed and compared with emissions arising from similar systems based on fossil fuels.
- Comparison from a life cycle perspective of the different alternatives for processing category 1 and category 2 animal by-products as discussed in section 9.2.2. Centralised and on-farm decentralised options of anaerobic digestion and other possible options could be of interest. A detailed comparison would include the modelling of the environmental burden (or credits) of the treatment processes and the final use of the different outputs of each alternative (for example in the case of anaerobic digestion: including the digestion process and the final use of biogas and digestate produced).
- Simultaneous modelling of the GHG emissions of all the systems affected by changes in the destiny or final use of animal by-products. This would potentially include: (i) production of various farmed animal systems, (ii) meat processing, (iii) production of marginal fats and proteins, (iv) production of oleochemicals, (v) rendering, (vi) petfoods manufacturing, (vii) production of animal feeds, (viii) alternative disposal options for on-farm mortalities, (ix) several alternatives for production of bioenergy, and any other systems that may be interlinked. Modelling the GHG emissions of such a system would help to identify what the ideal use is of animal by-products derived biomass. Such a model could be developed in a way that it could support the analysis of issues not only related to animal by-products but in general to animal derived food systems.

Chapter 10

Conclusions

10 Conclusions

- Mammalian derived rendered products in the UK have lower CO_2 emissions compared to substitute vegetable oils and protein meals, because mammalian ABP were treated as wastes within LCA methodology. Consequently, they do not carry any of the environmental burden associated with their production. In addition, a significant percentage of the thermal energy required for rendering was derived from combustion of MRF that does not contribute to fossil CO_2 emissions. The system also gains credits from the production of biogenic electricity from the combustion of MBM.
- The inclusion of animal slaughtering and ABP rendering in the system boundaries of meat processing does not radically affect the GHG emissions associated with meat production. The treatment of ABPs by rendering provides credits to the meat production system through the avoidance of products such as fossil fuels, and marginal protein meals and fats.
- The use of different co-product handling approaches during poultry processing and poultry ABP processing produces different GHG emissions for different edible co-products and poultry rendered products. Provided economic allocation is used during poultry processing to allocate GHG emissions between edible co-products and by-products, the GHG emissions of poultry rendered products such as PPAP, PRF and HFM are lower than those of alternatives such as soya bean meal and soya oil. Using this co-product handling approach, the GHG emissions of poultry rendered products depends largely on the relative proportion of category 1 rendered fat and natural gas used as fuels during poultry ABP processing.
- Allocation is essential in attributional studies where results are required for all the co-products and by-products produced. When the objective of the study is to investigate the environmental burden of the whole food system, the system should be expanded to include the effects associated with disposal of unavoidable by-products.

- The inclusion of animal by-product derived ingredients in salmonid feed formulations results in lower GHG emissions on a feed mass produced basis when they substitute part of vegetable or small pelagic species fish derived ingredients. These findings are strongly dependent on the co-product handling method used to partition between livestock co-products and by-products. GHG emissions of feed formulations including land based animal by-products are reduced when rendered fats are used as fuel for rendering because of the emissions credits associated with the category 1 rendering system. Economic allocation is perceived as an appropriate co-product handling approach for this type of system since the need for marginal feed ingredients is driven by the global expansion of the livestock production sector.
- Results for GHG emissions of rendered products are strongly dependent on the way their production is ascribed (co-production handling used to divide the inputs and outputs between animal edible products and by-products) and the GHG emission intensity of the energy carriers used in their production.

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Appendices

Appendix A

Table A-1 Greenhouse gas emissions results and some methodological characteristics of beef production systems studies

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Subak, 1999)	United States and Africa	1 kg CW US yield 54% Sahelian yield 61%	Literature	Main product	Y	Y	Y	Y	Y	N	Y	NA	NA	NA	NA	US feetlot 14.8 kg CO_2e Sahelian pastoral 8.4 kg CO_2e
(Phetteplace *et al.*, 2001)	United States	1 kg live weight gain per year for different type of operations	National Statistics, simulation, literature	Not required	Y	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Cow-calf 20.6±3.9kg CO_2e Stocker 14.4±2.3kg CO_2e Feedlot 5.66±0.24kg CO_2e Cow-calf through feedlot 15.5±2.3kg CO_2e
(Williams *et al.*, 2006)	England and Wales	1 tonne CW 55% yield (here presented on kg)	National statistics, literature and databases. Based on system modelling.	Economic allocation Relation between dairy and beef is included in the model System expansion	Y	Y	Y	Y	Y	Y	N	NA	NA	NA	NA	Weighted average 16 kg CO_2-eq Non-organic 15.8 kg CO_2e Organic 18.2 kg CO_2e 100% suckler

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	EF	EU	MM	SM	CG	LT	TA	S	C	WM	
				for manure fertilising												25.5 kg CO_2e Lowland 15.6 kg CO_2e Hill & upland 16.4 kg CO_2e Average based on LW 6.888 kg CO_2e (pers. comm. Williams 2011)
(Casey and Holden, 2006a)	Ireland	1 kg LW per year	Real farm data Literature and databases	Feeds are from databases that sometimes require co-product handling	Y	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Conventional 13 kg CO_2e Agri-environmental 12.2 kg CO_2e Organic 11.1 kg CO_2-eq
(Casey and Holden, 2006b)	Ireland	1 kg LW per year	National statistics Literature and databases	Mass allocation milk and meat Feeds are from databases that sometimes require co-product	Y	Y	Y	Y	Y	N	NR	NA	NA	NA	NA	11.26 kg CO_2-eq

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	EF	EU	MM	SM	CG	LT	TA	S	C	WM	
				handling												
(Ogino *et al.*, 2007a)	Japan	1 kg CW 40% yield	National statistics literature databases	Meat output: main product Feed not very clear, but it seem from databases	Y	Y	Y	Y	N	N	NR	NA	NA	NA	NA	36.4 kg CO_2-eq
(Vergé *et al.*, 2008)	Canada	1 kg LW	National statistics Literature databases	Main product for feeds	Y	Y	Y	Y	Y	Y	NR	NA	NA	NA	NA	10.37 kg CO_2-eq
(Weidema *et al.*, 2008b)	Europe	1 kg slaughter weight	Input-output tables, National Statistics, process data (hybrid), Databases	Economic allocation	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	Y	28.7 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Edward-Jones *et al.*, 2009)	Wales	1 kg LW	Real farm data Literature databases	Economic allocationbetween lamb, beef, cull ewes and Wool Feed Not clear if it was required for feeds	Y	Y	Y	Y	Y	N	NR	NA	NA	NA	NA	Conventional 9.7 – 38.1 kg CO_2-eq Extensive 18.8 – 132.6 kg CO_2-eq
(Cederberg *et al.*, 2009a)	Sweden	1 kg CW (based on statistics associated with CW)	National statistics, industry, literature, databases hybrid	Allocation between milk and beef based on feed requirements to cover the dairy cow´s milk production, maintenance and pregnancy Feeds: economic allocation When manure is a by-product: with system expansion	Y	Y	Y	Y	Y	N	N	N*	N*	NA	NA	19.8 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Nguyen *et al.*, 2010a)	Europe	1 kg CW Suckler cow-calf 51 – 58% Yield	European level data Literature databases	Allocation between milk and beef based on feed requirements to cover the dairy cow´s milk production, maintenance and pregnancy Soya bean meal – oil with system expansion	Y	Y	Y	Y	Y	N	Y	N	NA	NA	NA	Suckler cow-calf 27.3 kg CO_2-eq 84.1 kg CO_2-eq (including land transformation) Dairy bull calf 12 months 16.0 kg CO_2-eq 62.4 (including land transformation) Dairy bull calf 16 months 17.9 kg CO_2-eq 63.6 (including land transformation) Dairy bull calf 24 months 19.9kg CO_2-eq 69.5 (including land transformation)
(Peters *et al.*, 2010)	Australia	1 kg CW 53% yield	Real farm, national level statistics and literature hybrid	Mass allocation	Y	Y	Y	Y	Y	N	NR	Y	Y	NA	NA	Organic system in Victoria 8.2 – 11.5 kg CO_2-eq Premium system in New South Wales 9.8 – 10.2 kg CO_2-eq Grain finished

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
																9.9 kg CO_2-eq Grass Finished 12 kg CO_2-eq
(Beauchemin *et al.*, 2010)	Western Canada	1 kg CW 60% yield	Farm simulation national statistics and literature	Main product for carcass yield Feeds are produced on farm so not required	Y	Y	Y	Y	Y	N	NR	NA	NA	NA	NA	CW 21.73 kg CO_2-eq LW 13.04 kg CO_2-eq
(Pelletier *et al.*, 2010b)	Upper Midwestern United States	1 kg LW	Real farm data Literature and databases	Gross energy content	Y	Y	Y	Y	Y	N	NR	NA	NA	NA	NA	Feedlot 14.8 kg CO_2-eq Backgrounding/Feedlot 16.2 kg CO_2-eq Pasture 19.2 kg CO_2-eq
(Cederberg *et al.*, 2011)	Brazil	1 kg CW (based on statistics associated with carcass	National statistics, literature, databases	Main product Land transformation	Y	Y	Y	Y	Y	N[*]	Y	N[c]	N[c]	NA	NA	Newly deforested land 726 kg CO_2-eq Legal Amazon Region

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	EF	EU	MM	SM	CG	LT	TA	S	C	WM	
		weight, -global average of carcass produced by cattle head)		emissions between timber and pasture land based on carbon content and logging data No feedlots, only pastures												180 kg CO_2-eq Average Brazil 44 kg CO_2-eq
(Foley *et al.*, 2011)	Ireland	1 kg CW	Farm simulation, national statistics, databases	Main product Feeds: databases with sometimes already are allocated	Y	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Average based in National Farm Survey 23.1 kg CO_2e Steer Moderate. 19.7kg CO_2e Steer Intensive. 22.0 kg CO_2e Bull Moderate. 18.9 kg CO_2e Bull Intensive 20.4kg CO_2e
(Beauchemin *et al.*, 2011)	Canada	1 kg CW	Farm simulation national statistics	Not required Main product for carcass yield	Y	Y	Y	Y	Y	N	Y	NA	NA	NA	NA	CW 19.89 - 23.14kg CO_2e LW

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and greenhouse gas (GHG) emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
			and literature													11.99 – 13.88 kg CO_2e
(Veysset *et al.*, 2010)	France	1 tonne LW over a year (here presented on kg)	Farm simulation national statistics and literature	Main product / not required	Y	Y	Y	Y	Y	Y	N	NR	NR	NA	NA	calf-to-weanling and fattened females 16.6 kg CO_2e calf-to-weanling 100% grassland farm 17.1 kg CO_2e calf-to-beef Beef steers production 14.9kg CO_2e
(Lesschen *et al.*, 2011)	Europe	1 kg edible beef defined as 0.9 of CW Yield 58%	European and national statistics, simulation and literature	Mature dairy cows were attributed to the dairy cow sector, whereas GHG emissions related to calves and heifers were attributed to the beef sector Feed from seeds seem based on mass or some simple approach	Y	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Average EU-27 22.6 kg CO_2e Range for different countries ~17 - ~42 kg CO_2e

Study	**Geographical Coverage**	**Functional unit (FU)**	**Approach to economic and natural flow calculation**	**Co-product handling**	**Scope and greenhouse gas (GHG) emissions sources**											**GHG per FU**
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Eady *et al.*, 2011)	Australia	1 kg LW	Real farm data, national statistics literature, databases	Economic allocation	NR	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Gympie 17.5 - 22.9 kg CO_2e Arcadia Valley 11.6 to 15.5 kg CO_2e
(Capper, 2011)	United States	1 billion kg (here presented on kg)	National statistics literature, databases	Biological allocation between beef and dairy	Y	Y	Y	Y	Y	N	N	N	N	N	N	17.945 kg CO_2e

Y: Yes, it is included
N: No, it is no included
NR: not required
NA: not applicable
F: feed (production and delivery if required)
MM: manure management (direct and/or indirect)
SM: soil management (direct and/or indirect)
EF: enteric fermentation
EU: energy use (on-farm)
TA: transport of finished animals from farm to slaughtering (not necessarily required in every study, some studies may include transport of animal between farms)
CG: capital goods (production and maintenance)
LT: Land use change transformation (only required for marginal production)
S: slaughtering (inputs and outputs)
C: Consumption
WM: Final waste management
CW: carcass weight
LW: live weight
N[c:] Energy and capital goods for slaughtering and transport to Europe included in associated extended document (Cederberg *et al.*, 2009b)

Table A-2 Greenhouse gas emissions results and some methodological characteristics of lamb production systems studies

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	EF	EU	MM	SM	CG	LT	TA	S	C	WM	
Williams *et al.*(2006)	England and Wales	1 tonne CW 47% yield (here presented on kg)	National statistics, literature and databases. Based on system modelling.	Economic allocation between mutton and meat. System expansion for manure fertilising	Y	Y	Y	Y	Y	Y	N	NA	NA	NA	NA	Weighted average 17 kg CO_2-eq Non-organic 17.5 kg CO_2e Organic 10.1 kg CO_2e Average based on LW 7.501 kg CO_2e (pers. comm. Williams 2011)
Edward-Jones *et al.* (2009)	Wales	1 kg LW	Real farm data Literature databases	Economic allocationbetween lamb, beef, cull ewes and Wool Feed Not clear if it was required for feeds	Y	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Conventional 8.1 – 31.7 kg CO_2-eq Extensive 20.3. – 143.5 kg CO_2-eq

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources											GHG per FU
					Cradle-to-grave											
					Cradle-to-slaughterhouse-gate											
					Cradle-to-farm-gate											
					F	**EF**	**EU**	**MM**	**SM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Peters *et al.*, 2010)	Australia	1 kg CW 47% yield	Real farm, national level statistics and literature hybrid	Mass allocation	Y	Y	Y	Y	Y	N	N	Y	Y	NA	NA	8.3 – 7.2 kg CO_2-eq
(Biswas *et al.*, 2010)	Australia	1 kg LW	Real farm data Literature	Economic allocation for lamb and wool	Y	Y	Y	Y	Y	Y	N	NA	NA	NA	NA	Sub-clover 5.56 Mixed pasture 5.09

Y: Yes, it is included
N: No, it is no included
NA: not applicable
F: feed (production and delivery if required)
MM: manure management (direct and/or indirect)
SM: soil management (direct and/or indirect)
EF: enteric fermentation
EU: energy use (on-farm)
TA: transport of finished animals from farm to slaughtering (not necessarily required in every study, some studies may include transport of animal between farms)
CG: capital goods (production and maintenance)
LT: Land use change transformation (only required for marginal production)
S: slaughtering (inputs and outputs)
C: Consumption
WM: Final waste management
CW: carcass weight
LW: live weight

Table A-3 Greenhouse gas emissions results and some methodological characteristics of pig production systems studies

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources										GHG per FU
					Cradle-to-grave										
					Cradle-to-slaughterhouse-gate										
					Cradle-to-farm-gate										
					F	EF	EU	MM	CG	LT	TA	S	C	WM	
(Cederberg and Flysjö, 2004)	Sweden	1 kg bone and fat free meat (58%-59% CW)	National statistics Industry data, expert judgement,	Meat: primary product Feeds: economic allocation	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Animal welfare 4.08 kg CO_2e Environmental 3.63 kg CO_2e Product quality 4.43 kg CO_2e
(Basset-Mens and van der Werf, 2005)	France	1 kg LW	Generic management guidance, Industry data literature and databases.	Economic allocation between porker and sow	Y	Y	Y	Y	Y	N	NA	NA	NA	NA	Good Agricultural Practices (conventional) 2.3 kg CO_2e Red Label quality 3.46 kg CO_2e Organic 3.97 kg CO_2e
(Williams *et al.*, 2006)	England and Wales	1 tonne CW 72% 75% 77% yield (here presented on kg)	National statistics, literature and databases. Based on system modelling.	System expansion for manure fertilising	Y	Y	Y	Y	Y	N	NA	NA	NA	NA	Weighted average 6.4 kg CO_2e Non-organic 6.3kg CO_2e Organic 5.6 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources										GHG per FU
					Cradle-to-grave										
					Cradle-to-slaughterhouse-gate										
					Cradle-to-farm-gate										
					F	EF	EU	MM	CG	LT	TA	S	C	WM	
															Heavier finishing 6.1 kg CO_2e Indoor breeding 6.4 kg CO_2e Average based on LW 3.027 kg CO_2e (pers. comm. Williams 2011)
(Dalgaard *et al.*, 2007)	Denmark	1 kg CW (average yield 75%)	Generic practices, industry data, literature, databases	Feed: system expansion	Y	Y	Y	Y	N	N	Y	Y	NA	NA	3.6 kg CO_2e
(Weidema *et al.*, 2008b)	Europe	1 kg slaughter weight	Input-output tables, National Statistics, process data (hybrid), Databases	Economic allocation	Y	Y	Y	Y	Y		Y	Y	Y	Y	11.2 kg CO_2e
(Verge *et al.*, 2009a)	Canada	1 kg LW	National statistics Literature databases	Main product for feeds	Y	Y	Y	Y	Y		NA	NA	NA	NA	2.31 kg CO_2e

Study	**Geographical Coverage**	**Functional unit (FU)**	**Approach to economic and natural flow calculation**	**Co-product handling**	**Scope and GHG emissions sources**										**GHG per FU**
					Cradle-to-grave										
					Cradle-to-slaughterhouse-gate										
					Cradle-to-farm-gate										
					F	**EF**	**EU**	**MM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Cederberg *et al.*, 2009a)	Sweden	1 kg CW (based on statistics associated with CW)	National statistics, industry, literature, databases	main product Feeds: economic allocation When manure is a by-product: with system expansion	Y	Y	Y	Y	N	N	NA	NA	NA	NA	3.4 kg CO_2e
(Pelletier *et al.*, 2010a)	United States	1 kg LW	General practice, Industry surveys, literature, databases	Gross energy content	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Commodity 2.47 – 3.05 kg CO_2e Niche 2.52 – 3.33 kg CO_2e
(Nguyen *et al.*, 2010b)	Europe	1 kg CW 75% yield	European level data Literature databases	System expansion for soya bean meal and fish meal systems	Y	Y	Y	Y	N	Y	NA	NA	NA	NA	4.812 kg CO_2e With land transformation 9.752 kg CO_2e With land transformation and opportunity cost of capital 20.798 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources										GHG per FU
					Cradle-to-grave										
					Cradle-to-slaughterhouse-gate										
					Cradle-to-farm-gate										
					F	**EF**	**EU**	**MM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
(Lesschen *et al.*, 2011)	Europe	1 kg edible beef defined as 0.9 of CW Yield 75%	European and national statistics, simulation and literature	Main product	Y	Y	Y	Y	N	N	NA	NA	NA	NA	Average EU 3.5 kg CO_2e Range for different countries ~2.3 - ~7.3 kg CO_2e
(Phong *et al.*, 2011)	Vietnam	1 kg LW	Real farm data, literature,	Economic allocation between: Pig, Poultry/eggs, Fish, Rice grain, Fruits, Vegetables Economic allocation	Y	Y	Y	Y	N	N	NA	NA	NA	NA	8.262 kg CO_2e
(Stone *et al.*, 2012)	United States	1 head of swine (here presented on kg LW)	Generic modern process, industry data, expert judgement, literature, databases	Feed ingredients from databases already allocated, the author provides different allocation scenarios	Y	Y	Y	Y	N	N	NA	NA	NA	NA	4.47 kg CO_2e

Y: Yes, it is included
N: No, it is no included
NA: not applicable
F: feed (production and delivery if required)
MM: manure management (direct and/or indirect)
SM: soil management (direct and/or indirect)

EF: enteric fermentation
EU: energy use (on-farm)
TA: transport of finished animals from farm to slaughtering (not necessarily required in every study, some studies may include transport of animal between farms)CG: capital goods (production and maintenance)
LT: Land use change transformation (only required for marginal production)
S: slaughtering (inputs and outputs)
C: Consumption
WM: Final waste management
CW: carcass weight
LW: live weight

Table A-4 Greenhouse gas emissions results and some methodological characteristics of poultry production systems studies

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources									GHG per FU
					Cradle-to-grave									
					Cradle-to-slaughterhouse-gate									
					Cradle-to-farm-gate									
					F	EU	MM	CG	LT	TA	S	C	WM	
(Williams *et al.*, 2006)	England and Wales	1 tonneCW 70% yield (here presented on kg)	National statistics, literature and databases. Based on system modelling.	Economic allocation System expansion for manure fertilising	Y	Y	Y*	Y	N	NA	NA	NA	NA	Weighted average 4.5 kg CO_2e Non-organic 4.5 kg CO_2e Organic 6.7 kg CO_2e Free range 5.5 kg CO_2e Average based on LW 2.016 kg CO_2e (pers. comm. Williams 2011)
(Pelletier, 2008)	United States	1 tonne LW (here presented on kg)	Real farm data Expert judgement Literature Databases	Output LW not required Feeds: gross energy content Manure: system expansion	Y	Y	Y	N	N	NA	NA	NA	NA	1.395 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources									GHG per FU
					Cradle-to-grave									
					Cradle-to-slaughterhouse-gate									
					Cradle-to-farm-gate									
					F	EU	MM	CG	LT	TA	S	C	WM	
(Weidema *et al.*, 2008b)	Europe	1 kg slaughter weight	Input-output tables, National Statistics, process data (hybrid), Databases	Economic allocation	Y	Y	Y	Y	Y	Y	Y	Y	Y	3.6 kg CO_2e
(Verge *et al.*, 2009b)	Canada	1 kg LW	National statistics Literature databases	GHG emissions from culled layer hens is ascribed to eggs Main product for feeds	Y	Y	Y	Y	N	NA	NA	NA	NA	Broilers and culled layers 1.00 kg CO_2e Turkeys 1.44 kg CO_2e
(Cederberg *et al.*, 2009a)	Sweden	1 kg CW (based on statistics associated with CW) 70% yield	National statistics, industry, literature, databases	Poultry: main product Feeds: economic allocation When manure is a by-product: with system expansion	Y	Y	Y	N	N	NA	NA	NA	NA	1.9 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources									GHG per FU
					Cradle-to-grave									
					Cradle-to-slaughterhouse-gate									
					Cradle-to-farm-gate									
					F	EU	MM	CG	LT	TA	S	C	WM	
(Boggia *et al.*, 2010)	Italy	1 kg of LW	Real farm data Literature databases	Output: LW not required Feed are from databases that do have allocation	Y	Y	Y	Y	N	NA	NA	NA	NA	Conventional 0.689 kg CO_2e Organic 0.658 kg CO_2e Organic plus 0.703 kg CO_2e GHG has been calculated as non-biogenic CO_2 emission + 25*CH_4 emission
(Lesschen *et al.*, 2011)	Europe	1 kg edible beef defined as 0.9 of CW Yield 71%	European and national statistics, simulation and literature	GHG emissions from laying hens were attributed to eggs, whereas GHG emissions from broilers and other poultry were attributed to poultry Feed from seeds seem based on mass or some simple approach	Y	Y	Y	N	N	NA	NA	NA	NA	Average EU 1.7 kg CO_2e Range for different countries ~0.1 - ~4.2 kg CO_2e
(Phong *et al.*, 2011)	Vietnam	1 kg LW	Real farm data, literature,	Economic allocation between: Pig, Poultry/eggs, Fish, Rice grain, Fruits, Vegetables	Y	Y	Y	N	N	NA	NA	NA	NA	8.719 kg CO_2e

Study	Geographical Coverage	Functional unit (FU)	Approach to economic and natural flow calculation	Co-product handling	Scope and GHG emissions sources									GHG per FU
					Cradle-to-grave									
					Cradle-to-slaughterhouse-gate									
					Cradle-to-farm-gate									
					F	**EU**	**MM**	**CG**	**LT**	**TA**	**S**	**C**	**WM**	
				Economic allocation										
(Leinonen *et al.*, 2012)	United Kingdom	1 tonne CW 70% yield (here presented on kg)	National statistics, literature and databases. Based on system modelling (Williams *et al.*, 2006)	Economic allocation System expansion for manure fertilising	Y	Y	Y*	Y	N	NA	NA	NA	NA	Standard 4.41 ± 0.44 kg CO_2e Free range 5.13 ± 0.52 kg CO_2e Organic 4.41 ± 0.62 kg CO_2e

Y: Yes, it is included
N: No, it is no included
NA: not applicable
F: feed (production and delivery if required)
MM: manure management (direct and/or indirect)
SM: soil management (direct and/or indirect)
EU: energy use (on-farm)
TA: transport of finished animals from farm to slaughtering (not necessarily required in every study, some studies may include transport of animal between farms)
CG: capital goods (production and maintenance)
CG: capital goods (production and maintenance)
LT: Land use change transformation (only required for marginal production)
S: slaughtering (inputs and outputs)
C: Consumption
WM: Final waste management
CW: carcass weight
LW: live weight

Appendix B

Table B-1 Data sources used in the study

Section	Life cycle inventory	
	Main economic flows	**Secondary economic flows**
Rendering process category 1 and 3 (including administrative operations) (UK) See Fig 1	Yields: collected through direct contact with 7 UK rendering plants.	n.a.
	Thermal energy: 1. The amount of fuel consumed was collected through direct contact with UK rendering plants. 2. The amount of energy derived from mammalian rendered fat (MRF) was calculated using the equation: $Energy_{fuel} = m_{fuel} \times CV_{fuel}$ Where, m_{fuel}= mass of the fuel CV_{fuel}=the calorific value of the fuel. The Gross Calorific Value (GCV) of MRF (39 MJ/kg) was provided by a rendering company. The Net Calorific Value (NCV) of MRF (36 MJ/kg)was calculated using the equation (Lopez *et al.*, 2010): $GCV_{fuel} = 212.2\%H_{fuel} \times NCV_{fuel}$ Where, $\%H_{fuel}$ = the percentage of H by weight of the fuel. The %H of MRF (11%) was provided by a rendering company. 3. The amount of energy derived from natural gas was collected through direct contact with UK rendering plants in units of MWh. It was assumed that this was the net calorific energy as calorific value of gaseous fuels is very variable. The energy density of the life cycle inventory of natural gas in UK by Ecoinvent is in agreement with typical figures for calorific content of natural gas in UK. Note: The average amount of energy derived from natural gas and MRF was calculated by adding the amount of energy derived from natural gas to the amount of energy derived from MRF. This average does not take into account the combustion efficiency for the different fuels.	Natural gas: Ecoinvent database(Ecoinvent Centre, 2010) Category 1 MRF: produced by the system Biogenic CO_2 emissions were calculated according to the equation detailed below and assuming all the carbon contained in MRF is completely oxidised to CO_2. The Carbon Content of MRF (75%) was provided by a rendering company. $m_{CO2} = \frac{m_f \times \%C_f \times 44}{12}$ Where, m_{CO2} = amount of biogenic CO_2 emissions. m_f = mass of biogenic fuel burnt. $\%C_f$ = the % C by weight of the fuel.
	Electricity input: collected through direct contact with 5 UK rendering plants.	Ecoinvent database(Ecoinvent Centre, 2010)
	Wastewater output: collected through direct contact with 4 UK rendering plants.	Ecoinvent database(Ecoinvent Centre, 2010)
	Chemicals input: collected through direct contact with 4 UK rendering plants.	Ecoinvent database(Ecoinvent Centre, 2010)
	Water input: collected through direct contact with 4 UK rendering plants.	Ecoinvent database(Ecoinvent Centre, 2010)

Table B-1 (continuation) Data sources used in the study

Section	Life cycle inventory	
	Main economic flows	**Secondary economic flows**
Power from fluidized bed combustion with steam turbine plant (UK) for system expansion category 1 MBM	MBM input: collected through direct contact with 5 category 1 rendering plants.	n.a.
	Electricity output: calculated using the equation (Yassin *et al.*, 2009): $$E_{electricity} = \eta_{FBC} \times m_{MBM} \times NCV_{MBM}$$ Where, η_{FBC} =conversion efficiency FBC plants. m_{MBM}=the mass of the MBM burnt. NCV = the Net Calorific Value of the MBM. The conversion efficiency of FBC plants was 18% (Yassin *et al.*, 2009) The Gross Calorific Value of MBM (19.75 MJ/kg)was provided by companies. The Net Calorific Value of MBM (18.5 MJ/kg)was calculated using the equation (Lopez *et al.*, 2010): $$GHV_{fuel} = 212.2\%H_{fuel} \times NCV_{fuel}$$ Where, $\%H_{fuel}$ =the percentage of H by weight of the fuel. The %H of MBM (6%) was provided by a rendering company.	MBM: produced by the system Biogenic CO_2emissions were calculated according to the equation detailed below and assuming all the carbon contained in MBM is completely oxidised to CO_2. The carbon content of MBM (40%) was provided by a rendering company. $$m_{CO2} = \frac{m_f \times \%C_f \times 44}{12}$$ Where, m_{CO2} = amount of biogenic CO_2emissions. m_f = mass of biogenic fuel burnt. $\%C_f$ = the % C by weight of the fuel. Avoidance of electricity from the national grid: Ecoinvent database(Ecoinvent Centre, 2010)
	Ash amount to landfill: calculated using the percentage of material in a FBC plant that is obtained as ash. The percentage (15%) was provided through direct contact with 1 UK FBC plant.	Ecoinvent database (Ecoinvent Centre, 2010)
Transport (EU)	Transport distances of200 km by lorry were assumed, which is reasonable for UK road distances	Ecoinvent database (Ecoinvent Centre, 2010)
Soybean meal for system expansion through avoidance of PAP	Assumed to be equal to the amount of PAP produced.	Ecoinvent database (Ecoinvent Centre, 2010)

Appendix C

The potential for energy self-sufficiency in the United Kingdom rendering industry

The content of this appendix can be accessed as:

Ramirez AD, Humphries AC, Woodgate SL, Wilkinson RG (2011) The potential for energy self-sufficiency in the United Kingdom rendering industry. *Management of Natural Resources, Sustainable Development and Ecological Hazards III*. Transactions: Ecology and the Environment volume 148. Ravage of the Planet 2011 Conference proceedings Edited By: C.A. Brebbia, S.S. Zubir

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http://library.witpress.com/pages/PaperInfo.asp?PaperID=22995

The potential for energy self-sufficiency in the United Kingdom rendering industry

C.1 Abstract

Animal by-products (ABPs) are co-products of meat production system that include on-farm fallen stock and slaughterhouse co-products that are not fit for human consumption (different types of fat, offal and bone). Slaughterhouse co-products represent between 30 to 50% of the life weight of farm animals. In the UK, fat, offal and bone are normally handled by rendering. Rendering is a process where ABP are sized and then dried to produce rendered products: tallow and a protein meal. Depending on the category of the ABP, rendered products can be used in pet foods manufacturing, the oleochemical industry and as biofuels (with and without further processing). The UK rendering industry uses both tallow and natural gas as fuel for heat production during the drying process. A study of UK rendering plants was undertaken to determine the total tallow production and the relative proportions of tallow and natural gas used as fuels. Data on fuel and energy use was collected from five rendering plants, representing 50% of the ABP processed in the UK. The results indicate that tallow use by the UK rendering industry ranged from 15 – 100% of total heat production with the remainder being derived from natural gas. When scaled up, it can be calculated that between 2006 and 2008 the UK rendering industry required around 5.7 PJ of heat per annum. During the same period the energy potentially available from rendered tallow (usable as biofuel) was 6.7 PJ. It can be concluded that potentially the UK rendering industry could be self-sufficient in energy use. However, use of tallow as a biofuel depends on the relative cost of natural gas compared to alternative markets for tallow.

C.2 Introduction

The rendering industry is accepted to play an important role in the sustainable food chain by transforming animal by-products (ABPs) into processed proteins in form of the Meat and Bone Meal (MBM) and Processed Animal Proteins (PAP) and rendered fats (tallow). These ABPs provide feedstock material for other industries that include pet food manufacturing, the oleochemical industry and the biofuel industry. ABPs are secondary products of the animal agriculture and meat industries. ABPs include hides, skins, hairs, feathers, hoofs, horns, feet, heads, bones, toe nails, blood, organs, glands, intestines, muscle and fat tissues, shells and whole carcasses (Meeker and Hamilton, 2006).

ABPs and their final uses are classified into 3 categories according to European Legislation (EC, 2002; 2009). Category 1 material includes animals infected or suspected of infection with a Transmissible Spongiform Encephalopathy, Specified Risk Material (SRM) which in the UK includes entire head, vertebral column, tonsils, spinal cord and intestines of ruminants, and entire bodies containing SRM. Category 1 materials must be destroyed by combustion or rendering. Category 1 rendered products can be used as biofuels. Category 2 materials are mostly on-farm mortalities, manure and digestive tract content. Category 2 rendered fats can be used in the oleochemical industry and for the production of fertilizers. MBM can be used as fertilizer or anaerobically digested to produce biogas. Category 3 materials are ABPs that are fit for human consumption or unfit but with no transmissible diseases. The final use of this material, besides those detailed for category 1 and 2, is as raw material for pet food manufacturing. When different category materials are mixed together they are classified as the lower category in the mix (EC, 2002; 2009). Normally in the UK, category 1 and 2 materials are mixed together and treated as category 1 materials.

According to Woodgate (pers comm 2010) the UK rendering industry processed over 2 million Mg of ABP between per annum between 2006 and 2008. The various uses for rendered protein and fat are presented in figures C-1 and C-2 respectively. Combustion as a fuel is the most important use for category 1 rendered fats. They are normally used as fuels for boilers in rendering plants. Biodiesel production is also an important use for every category of rendered fats.

Rendering is an energy intensive process that involves both physical and chemical transformation, with the processes involving particle size reduction, heat treatment (for dehydration and microbial sterilization), pressing, separation and milling (Woodgate and van der Veen, 2004). The combustion of fossil fuels mainly associated with energy use is the most important source of carbon dioxide.

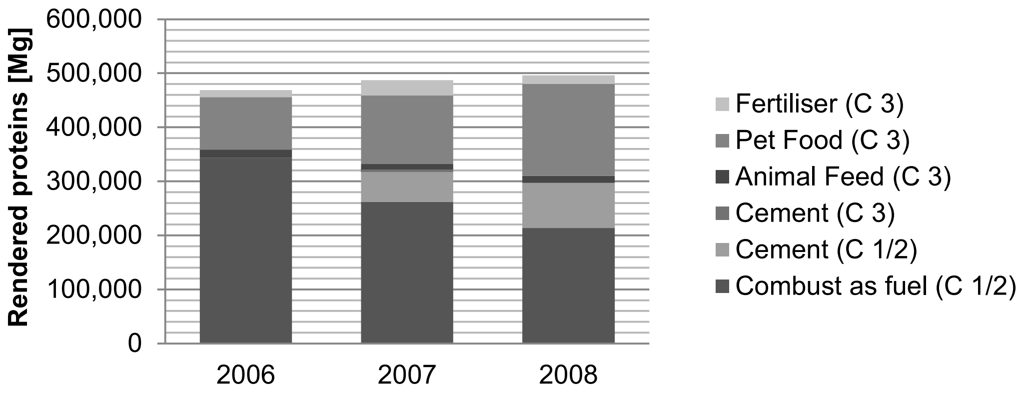

Figure C-1 Uses of rendered proteins in the UK between 2006 and 2008 Woodgate (pers comm 2010)

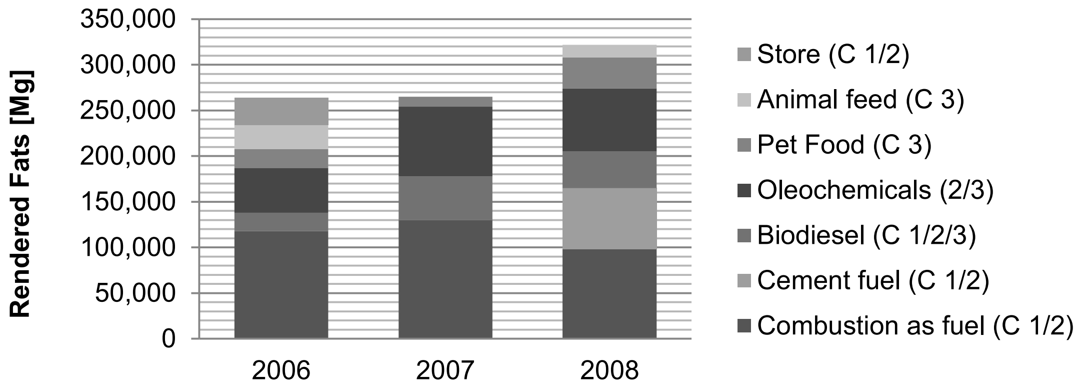

Figure C-2 Uses of rendered fats in the UK between 2006 and 2008 Woodgate (pers comm 2010)

The objective of this study was to estimate the consumption of energy by the UK rendering industry. Further objectives of the study were to estimate the percentage use of self-produced tallow by the UK rendering industry, and to analyse whether the UK rendering industry could be energy self-sufficient.

C.3 Methodology

C.3.1 Energy consumption in the UK rendering industry

Data from six UK rendering plants were collected directly through the completion of a structured questionnaire. Data was collected for the years 2006, 2007, and 2008. Specific data collected were ABP yield, the amount of energy consumed (i.e. electricity, tallow, and natural gas), and the energy content of tallow and MBM.

The yield of protein meals and tallow produced by each rendering plant in each of the three years studied was gathered. Yield is defined as the percentage (in mass) of ABP converted to rendered product, and can be considered indicative of the water content of the ABP to be processed. For example, the higher the water content of the ABP to be processed, the lower the yield due to the evaporation of water during the rendering process (i.e. mass is lost from the system).

The energy released from the combustion of fuel (natural gas or tallow) by the rendering plants studied was calculated using the gross calorific value of the fuel (Table C-1) and multiplying it by the annual amount of fuel used by each plant (Equation 1). Combustion efficiency was ignored as the purpose of the study was to estimate the energy used in fuels. The annual amount of fuel used was obtained directly from the plants records.

The expression to calculate the (annual) heat energy use is:

$$E_{heat\ x} = m_{fuel\ x} \times GCV_{fuel\ x} \quad (1)$$

Where

$E_{heat\ x}$ is the energy contained in the fuel *x*,

$m_{fuel\ x}$ is the mass of fuel *x* and,

$\boldsymbol{GCV_{fuel\ x}}$ is the gross calorific value of fuel x (Table C-1)

Table C-1 Gross calorific value of fuels

Fuel	GCV (MJ/kg)	Source
Tallow	39	Provided by plants
Natural gas	54	(NPL, 2008)

The annual amount of electricity consumed by each plant was obtained directly from records regarding the annual amount of electricity purchased.

C.3.2 Potential for energy self-sufficiency in the UK rendering industry

To estimate the annual amount of energy consumed for thermal energy production ($E_{thermal\text{-}required}$) by the UK rendering industry, the average amount of energy consumed by the six plants examined in the current study ($h_{rendering}$ MJ/kg ABP) was multiplied by the annual mass of ABP processed by the UK rendering industry ($m_{ABP-procesed}$) according to equation 2.

The amount of thermal energy that could be produced by the UK rendering industry ($E_{thermal\text{-}potential}$) from the combustion of tallow was calculated from the amount of category 1 tallow produced annually ($m_{cat1tallow-avai}$) according to equation 3. The potential for the UK rendering industry to be self-sufficient in terms of thermal energy demand was expressed as a ratio, referred to as the thermal self-sufficiency ratio ($SSR_{thermal}$), equation 4.

$$E_{thermal-required} = m_{ABP-procesed} h_{rendering} \quad (2)$$

$$E_{thermal-potential} = m_{cat1tallow-avai} GCV_{tallow} \quad (3)$$

$$SSR_{thermal} = \frac{E_{thermal-potential}}{E_{thermal-required}} \quad (4)$$

To estimate the annual amount of electricity consumed by the UK rendering industry ($E_{electric\text{-}required}$), the average amount of electricity consumed by the six plants examined in the current study ($e_{rendering}$in MJ/kg ABP) was multiplied by the annual mass of ABP processed by the UK rendering industry (Equation 5).

The amount of electricity that could be produced by the UK rendering industry through the combustion of category 1 MBM in FCB plants was calculated by multiplying the energy content of MBM by the annual amount of category 1 MBM available ($m_{MBM\text{-}avai}$) and by assuming efficiency of conversion (η_{FBC})(equation 6) an efficiency of 18% taken from Yassin *et al.* (2009)

$$E_{electric-required} = m_{ABP-procesed} e_{rendering} \quad (5)$$

$$E_{electric-potential} = \eta_{FBC} m_{MBM-avai} GCV_{MBM} \quad (6)$$

Where GCV_{MBM} is the Gross Calorific Value of MBM (19.8 MJ/kg as indicated by the rendering companies participating in the current study).

The potential for the UK rendering industry to be self-sufficient in terms of electricity demand was expressed as a ratio, referred to as the self-sufficiency ratio for electricity ($SSR_{electric}$) and according to equation 7.

$$SSR_{electric} = \frac{E_{electric-potential}}{E_{electric-required}} \quad (7)$$

C.4 Results and discussion

C.4.1 Energy consumption in the UK rendering industry

The six plants included in the current study processed between 40 and 50% of the annual ABPs processed in the UK between 2006 and 2008 and hence represent a significant proportion of the UK rendering industry. Five of the six plants processed mammalian material and only one processed poultry material. Yields varied between the different plants, although within plants they were reasonably stable throughout the three years of the study, Figure C-3. The average yield of the 6 plants was 29% protein meals and 15% tallow (mass of rendered products expressed as percentage of mass of ABP processed). Lopez *et al.* (2010) reported a yield of 23% protein meal and 28% tallow for beef ABP rendering in the US. The average yields detailed in the current study are significantly different to those reported in Lopez *et al.* (2010). This may be due to differences in the water content of the ABPs processed.

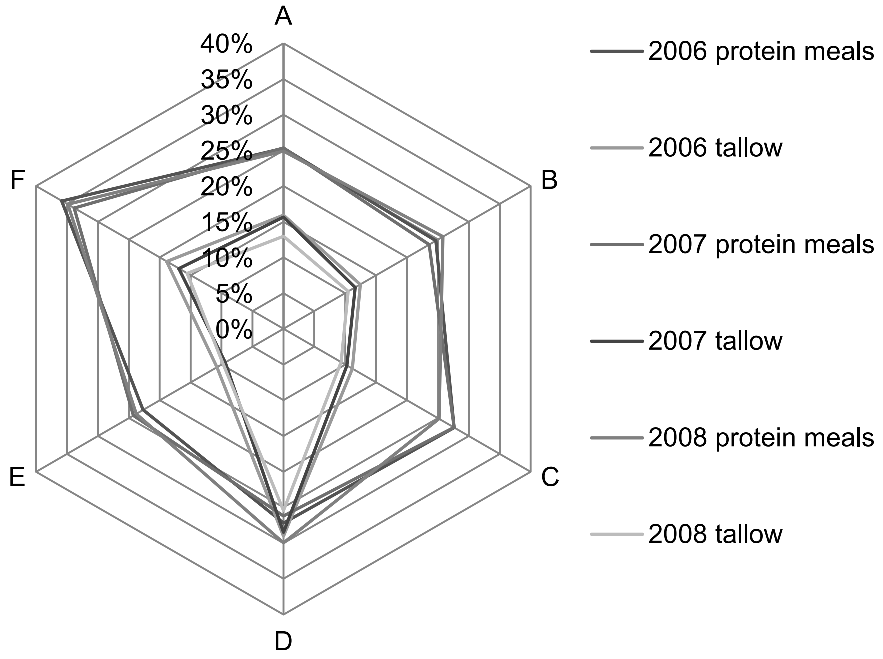

Figure C-3 Annual yields of meat and bone meal and tallow (% ABP processed) for 6 rendering plants (A to F) in the UK between the years 2006 and 2008

The relationship between the amount of ABP processed and energy consumption by the 6 rendering plants used in the study is presented in Figure C-4. There is a strong linear relationship between the amount of ABP processed and the energy contained in combustion fuels (R^2=0.95), and a fairly strong linear relationship between the amount of ABP processed and electricity consumption (R^2=0.72).

The amount of energy contained in fuels to produce thermal energy for the rendering process ranged between 1.4 and 3.4 MJ/kg ABP processed for the six plants studied. Similarly, the amount of electricity used by the six rendering plants ranged between 0.1 and 0.4 MJ/kg ABP processed. Research by Lopez *et al.* (2010) reported energy consumption by the US rendering industry of 2.8 MJ/kg ABP processed, of which 0.3 MJ/kg ABP was in the form of electricity and 2.5 MJ/kg ABP was provided from the combustion of fuels. In contrast, Ramírez *et al*. (2006) reported a figure for the European rendering industry of 1.6 MJ/kg ABP processed. The values reported by the current study are in reasonable agreement with the values provided by Lopez *et al.* (2010) and Ramírez *et al*. (2006).

Regarding percentage of heat and electricity, for the six rendering plants included in the current study, electricity accounted for approximately 8% of energy with the remaining attributable to thermal energy. These values are in agreement with the figures reported by Lopez *et al.* (2010) for the US rendering industry (9.4% for electricity).

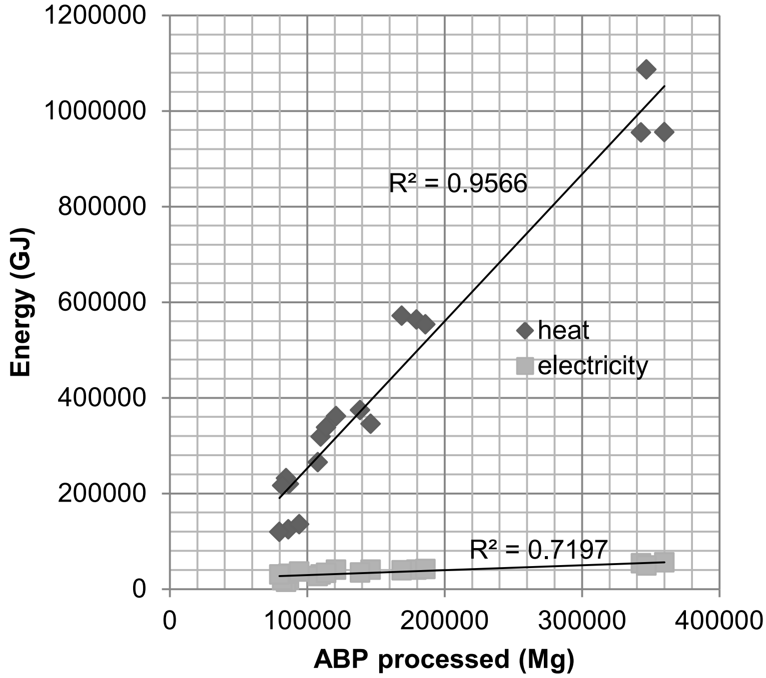

Figure C-4 Relationship between the amount of animal by-product processed and energy consumption by 6 UK rendering plants between the years 2006 and 2008

The percentage of total thermal energy contained in tallow and natural gas used for the rendering process varied between plants, and between years as illustrated in Figure C-5. For example, plants B and F used only tallow, whilst plants C and E used significantly less tallow in 2007 and 2008 compared to 2006. The reasons for these variations probably reflect the relative cost of natural gas compared to alternative markets for tallow. The weighted average proportion of thermal process energy derived from tallow for the six plants was 76%, with the remaining energy being derived from natural gas. Thus, the rendering industry already uses a relatively high proportion of thermal energy derived from tallow.

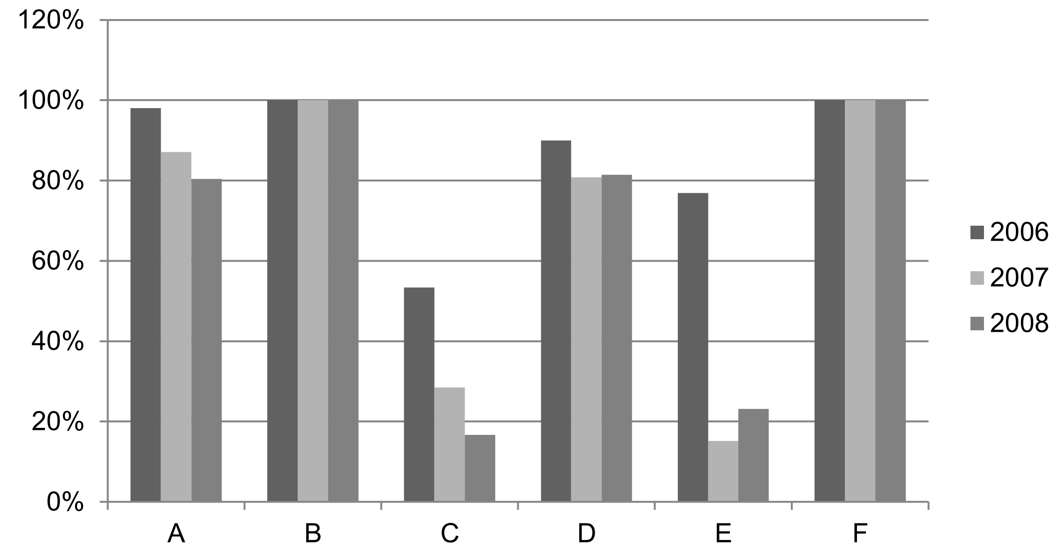

Figure C-5 Annual percentage of thermal energy consumption derived from tallow for 6 UK rendering plants between 2006 and 2008

C.4.2 Potential for energy self-sufficiency in the UK rendering industry

The thermal energy requirement and the potential thermal energy production from tallow and the thermal self-sufficiency ratios for the UK rendering industry are provided in Table C-2 for the years 2006, 2007, and 2008. The self-sufficiency ratios calculated are all greater than 1.0, indicating that the UK rendering industries thermal demand could be satisfied from the combustion of category 1 tallow. The generation of thermal energy from the combustion of category 1 tallow reduces the demand for finite fossil fuels, whilst additionally reducing greenhouse gas emissions because the biomass is associated with a short carbon cycle (Astrup *et al.*, 2009). However alternative uses of category 1 tallow (e.g. biodiesel production) should be analysed to ascertain which uses of tallow provides the highest climate and financial benefits.

Table C-2 The thermal energy requirement, the potential thermal energy production from category 1 tallow and the thermal self-sufficiency ratios for the UK rendering industry between 2006 and 2008

	2006	**2007**	**2008**
$E_{thermal-potential}$(PJ)	6.8	6.5	6.8
$E_{thermal-required}$(PJ)	5.6	6.1	5.5
$SSR_{thermal}$	1.2	1.1	1.2

The electricity requirement, and the potential electricity production from meat and bone meal and the electricity self-sufficiency ratios for the UK rendering industry are provided in Table C-3 for the years 2006, 2007, and 2008. The self-sufficiency ratios calculated are greater than 1, indicating that the rendering industry could be self-sufficient in terms of electricity. Electricity obtained from MBM has climate benefits in comparison to the use of British electricity, which is predominantly produced from the combustion of coal which is associated with high carbon emissions due to the properties of the fuel and the efficiency of the conversion process. MBM can also be used to produce heat in the cement industry; however as can be seen in fig. 3 this is not currently the main use. Greenhouse gas emissions from alternative uses of category 1 MBM should be analysed to ascertain which uses of category 1 MBM provides the highest climate and financial benefits.

Table C-3 The electricity requirement, the potential electricity production from category 1 MBM and the electricity self-sufficiency ratios for the UK rendering industry between 2006 and 2008

	2006	**2007**	**2008**
$E_{electric-potential}$(PJ)	1.2	1.1	1.1
$E_{electric-required}$(PJ)	0.5	0.5	0.4
$SSR_{thermal}$	2.7	2.3	2.3

C.5 Conclusion

The UK rendering industry produces different categories of rendered products. Under current EU legislation category 1 rendered products are required to be destroyed by combustion. During the years 2006 to 2008 sufficient quantities of category 1 rendered products were produced to satisfy both the thermal energy and electricity requirements of the UK rendering industry. Rendered products are produced from ABPs which are unavoidable by-products of livestock production and meat processing; consequently, they do not carry an associated environmental burden. The UK rendering industry could be self-sufficient in energy use by utilising category 1 rendered products as biofuels