REDUCING UNCERTAINTY IN LIFE CYCLE ASSESSMENT OF LIVESTOCK PRODUCTION SYSTEMS

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[37,334 words]

Abstract

Life cycle assessment (LCA) has been increasingly applied to livestock production systems to estimate their environmental footprints, but the degree of uncertainties associated with these values is known to be generally high. This thesis explores novel methods of LCA modelling to reduce uncertainty associated with environmental footprints of meat production systems, with the view to contribute to objective and transparent debates about the role of livestock in global food security. Three innovative approaches are proposed in this thesis. First, as information on individual animals is often unavailable, livestock data are often aggregated at the time of inventory analysis. To investigate the level of bias caused by this aggregation, Chapter 3 uses primary data collected at the North Wyke Farm Platform in Southwest England and calculates emission intensities for individual animals and their intra-farm distributions, providing a step towards deriving optimal animal selection strategies based on livestock LCA. Second, the severity of greenhouse gas emissions from agricultural production is known to vary spatially and temporally, yet available LCA frameworks often fail to sufficiently consider these differences due to data constraints. To evaluate the degree of avoidable uncertainties attributable to this practice, Chapter 4 conducts an original field experiment to derive site-specific nitrous oxide emission factors, which are subsequently used in Chapter 5 to compare LCA results derived under these localised values and generic alternatives intended for the widest possible users. Finally, while LCA results are typically communicated in the form of environmental burdens per output of mass, it is gradually becoming recognised that product quality also needs to be accounted for to truly understand the value of each farming system to society. Using data from seven livestock production systems encompassing cattle, sheep, pigs, and poultry, Chapter 6 develops new methods to incorporate nutritional values of meat products into livestock LCA.

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I declare that the work in this dissertation was carried out in accordance with the requirements of the University's Regulations and Code of Practice for Research Degree Programmes and that it has not been submitted for any other academic award. Except where indicated by specific reference in the text, the work is the candidate's own work. Work done in collaboration with, or with the assistance of, others, is indicated as such. Any views expressed in the dissertation are those of the author.

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List of acronyms

ADG: average daily gain
AP: acidification potential
AVG: average pig herd performance in Chapter 2
BOD: biochemical oxygen demand
BW: average body weight of an animal category
CED: cumulative energy demand
CP: crude protein
CF: carbon footprint
COD: chemical oxygen demand
CON+N: control plot with nitrogen in field experiment
CON-N: control plot without nitrogen in field experiment
CVD: cardiovascular disease
CW: carcase weight
D: dung plot in field experiment
DE: digestible energy
DHA: docosahexaenoic acid
DM: dry matter
DMI: dry matter intake
EF: emission factor
EF_1: percentage of nitrogen applied as fertiliser lost as nitrous oxide
EF_3: percentage of nitrogen applied as a weighted average of dung and urine lost as nitrous oxide
EF_D: percentage of nitrogen applied as dung lost as nitrous oxide
EF_U: percentage of nitrogen applied as urine lost as nitrous oxide
EP: eutrophication potential
EPA: eicosapentaenoic acid
FAO: Food and Agricultural Organisation of the United Nations
FCE/FCR: feed conversion efficiency/feed conversion ratio
FD: fossil depletion
FNI: Finnish nutrient index
FU: functional unit
FYM: farmyard manure
GHG: greenhouse gas
GWP: global warming potential
HS: high sugar grass monoculture farmlet on the North Wyke Farm Platform
IPCC: Intergovernmental Panel on Climate Change
LCA: life cycle assessment
LCC: life cycle costing
LCI: life cycle inventory analysis
LCIA: life cycle impact assessment
LU: land use
LUC: land use change
LW: liveweight
LWG: liveweight gain
MADF: modified acid detergent fibre
MC: Monte Carlo
ME: metabolisable energy
MUFA: monounsaturated fatty acids
NDS: nutrient density score
NWFP: North Wyke Farm Platform
PP: permanent pasture farmlet on the North Wyke Farm Platform

PUFA: polyunsaturated fatty acids
RDA: recommended daily allowance
RDI: recommended daily intake
RoI: Republic of Ireland
SAA: synthetic amino acids
SD: standard deviation
SFA: saturated fatty acids
T10: top 10 percent of pig herds in Chapter 2
T25: top 25 percent of pig herds in Chapter 2
U: urine plot treatment on the field experiment
UKNI: United Kingdom nutrient index
WC: white clover mixed sward on the North Wyke Farm Platform
WFPS: water filled pore space
WSC: water soluble carbohydrate
Y_m: percentage of gross energy lost as methane

Publications resulting from this PhD research

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Peer reviewed journal articles (manuscripts in preparation)

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Chapter 1 – Introduction

1.1 Background

In order to support the projected global population of 9.15 billion people at mid-century, a 70% increase in total global food production is believed to be required (FAO, 2009) unless drastic measures are taken to improve global distribution of food (Ingram, 2011). In the context of animal production, outputs from meat and dairy enterprises worldwide must be increased, respectively, by at least 53% and 48% under current levels of food waste (Thornton, 2010), and possibly more if the FAO's nutritional recommendations for animal protein are proactively followed to address malnutrition, defined as both under nutrition (lack of food) and availability of micro-nutrients (hidden hunger) through a balanced diet (FAO, 2014). Worldwide livestock production, however, is estimated to generate 7.1 Gt CO_2-eq of greenhouse gases (GHG) each year, with beef and dairy cattle contributing 61%, followed by pigs, poultry (meat and eggs) and small ruminants at 9%, 8% and 6.5%, respectively (Gerber et al., 2013). The enormity of the sector's impact on climate change, accompanied by many equally important environmental issues caused by livestock—including but not limited to: eutrophication, soil compaction and erosion, and biodiversity losses—demonstrate the urgency of identifying economically and environmentally sustainable methods of livestock production which are critical to ensure long-term food security (Eisler et al., 2014).

With considerably lower environmental footprints associated with "white" meat produced from monogastric animals relative to ruminant livestock (de Vries and de Boer, 2010), improving the efficiency of pig production systems holds less potential to reduce climate impacts. Nevertheless, pig production has considerable detrimental effects to local soils and watercourses due to acidification resulting from ammonia (NH_3) emissions and eutrophication from nitrate (NO_3^-) and phosphate (PO_4^3) losses. Most intensive piggeries also demand vast amounts of human-edible crops, which account for around 64% of total feed (Wilkinson, 2011), that could potentially be used elsewhere to directly feed the human population (McAuliffe et al., 2016). Similarly, feedlot-based beef production systems, which tend to show a lower level of GHG emissions intensity than pasture-based systems (Pelletier et al., 2010b, Peters et al., 2010, Nguyen et al., 2010a), are known to be the least efficient users of cereals and legumes in the intensive agri-food industry, when compared to pigs (Steinfeld, 2006). Pastoral systems for ruminant production, on the other hand, can utilise land unsuitable for arable crop production by converting forages to valuable sources of protein and bioavailable micro-nutrients for humans (van Zanten et al., 2016) without driving the food-feed competition for resources (Wilkinson, 2011, Wilkinson and Lee, 2018). Given this apparent trade-off and the fact that expected population growth will likely increase demand for human-edible crops and animal-originated protein at the same time, improving the environmental and production efficiency of both monogastric and ruminant systems seems to form, at least for the foreseeable future, part of the solution package for the global issue of food security.

A primary means to evaluate environmental footprints of production systems is life cycle assessment (LCA) (de Vries et al., 2015). Although the method itself is applicable and indeed adopted by a wide range of industries far beyond agriculture, what separates agriculture from other industries, and in particular modern manufacturing, is the high degree of uncertainties associated with physical, chemical and biological processes that underpin its production systems. A case in point is described by Dudley et al. (2014), who examined differences in environmental performance between multiple US feedlot systems; using the Monte Carlo (MC) technique, the authors calculated global warming potential (GWP) at a range of 2.5 – 9.6 kg CO_2-eq/kg liveweight (LW). Such a wide confidence interval demonstrates the unreliable nature of point-estimates provided by LCA models that do not capture realistic variability (Chen and Corson, 2014). Furthermore, this also highlights the importance of efforts to improve our understanding of uncertainties through methodological development as well as more intensive data collection.

Uncertainties in LCA studies arise from various sources, of which perhaps the most important is the variability inherent in life cycle inventory analysis (LCI) data and in life cycle impact assessment (LCIA) methodology. The latter, however, has been shown to have negligible effects on a study's outputs once the timescale of analysis (e.g. 20 years, 100 years or 500 years) has been clearly defined in the case of GWP (Reckmann et al., 2013). Uncertainties associated with LCI, on the other hand, have considerable impacts on environmental footprints arising from different farming systems, and are also of stronger relevance to a large population of practitioners. Nonetheless, LCA studies often omit MC analysis (Imbeault-Tétreault et al., 2013), which is one of the most common approaches for assessing uncertainty in LCI along with scenario and sensitivity analyses (Curran, 2012). A recent review of uncertainty analysis (Igos et al., 2018) identified three levels of assessment within the LCA framework: (1) sensitivity and scenario analyses to examine the effects of individual parameters within a system (basic); (2) use of characterisation factors and analysis of expertly designed scenarios via MC (intermediate); and (3) careful consideration of correlations between variables and fundamental reasons behind uncertainties (advanced). Although some livestock LCAs consider uncertainty during model interpretation, the level of detail varies from study to study and, with the exception of a few studies (Leinonen et al., 2012, Mackenzie et al., 2015), seldom go further than what Igos et al. (2018) would have classified as "basic" analysis—as will be discussed in Section 1.5.

The overarching aim of this thesis, therefore, is to explore novel methods of LCA modelling to reduce uncertainty associated with environmental footprint of meat production systems and thereby contribute to objective and transparent debates about the role of livestock in global food security. Data will primarily be drawn from the North Wyke Farm Platform (NWFP), an intensively instrumented cattle-grazing farm located in southwest England, in order to accurately quantify the degree of uncertainty associated with different modelling strategies. However, pig systems will also be considered to account for environmental implications of monogastric animal production, and findings from both analyses will be integrated at the end to derive policy implications across the livestock industry.

1.2 Life cycle assessment methodology

The framework for carrying out an LCA is directed by International Organization for Standardization (ISO) 14040 guidelines (ISO, 2006). **Figure 1.1** outlines the four different phases of an LCA study as defined by Curran (2012). Each step is iterative and may require revisiting a previous phase as, for example, data availability changes or the goal of the study needs revision.

1.2.1 Goal and scope definition

The first step involves detailing the overall goal of the study. This includes identifying the audience and what will be achieved by the study. The scope of the study must be clearly defined by delineating system boundaries, such as from the extraction of raw materials to the farm gate, and choosing an appropriate functional unit (FU). The FU is the unit to which all burdens and benefits will be scaled; in the case of livestock LCA, for instance, the production of 1 kg of LW at the farm gate is one of the most frequently used FU. This unit should represent a realistic output and function from the system; other examples, therefore, include 1 kg of LW delivered to the slaughterhouse, 1 kg of carcase weight produced at the slaughterhouse and 1 kg of meat consumed by humans.

1.2.2 Life cycle inventory analysis

The LCI requires gathering input and output data for all relevant processes within the system boundary. These data may be for foreground processes (usually primary data from the system directly under examination) or background processes (usually secondary data pertaining to processes upstream and downstream from the foreground processes), and can be acquired from various sources. For example, data on transportation and associated emissions are available in the *ecoinvent* life cycle inventory database (Wernet et al., 2016) while those associated with crop production for concentrated feeds are found in agricultural databases, such as Agri-Footprint (Durlinger et al., 2017). Foreground data, usually representing the processes directly affected by decision making within a system, are generally gathered from surveys or examinations of local record. On the other hand, primary data for flows to nature (e.g. emissions from GHGs and losses of N and P) are either sourced from direct measurements or calculations, the latter often derived from Intergovernmental Panel on Climate Change (IPCC) guidelines (IPCC, 2006).

1.2.3 Life cycle impact assessment

Although impact categories studied vary depending on the goal of a study, GWP, acidification potential (AP) and eutrophication potential (EP) are included in many LCAs. Once the relevant inventory data have been collected, the next stage requires classifying emissions and losses to their relevant impact categories. For instance, NH_3 from pig slurry can contribute to soil acidification and freshwater eutrophication, and therefore, NH_3 emissions should be considered in both AP and EP impact categories. On the other hand, CO_2 emissions should be omitted from both the categories but considered for GWP. **Figure 1.2** demonstrates a simplified characterisation process (Baumann and Tillman, 2004) while **Table 1.1** presents impact factors adopted under the Leiden Institute of Environmental Sciences (CML) impact assessment (Heijungs et al., 1997). The LCIA stage quantifies the impact level of the data collected during LCI. Once the inventory has been specified by LCI, this process can be largely automated in practice using LCA software such as SimaPro (www.pre-sustainability.com) or GaBi (www.gabi-software.com).

1.2.4 Interpretation

The final stage of an LCA involves interpreting the results and communicating them appropriately to the intended audience. During this phase, collected LCI data are tested for accuracy and uncertainty through statistical analysis. A brief breakdown of these tests is displayed in **Table 1.2**. After the data have been tested, it must be determined how they will be presented. This will largely depend on the goal of the study, as LCA studies can generate conclusions and policy implications with varying complexity.

1.3 LCA applied to meat production systems

LCA has been applied to all four of the globally important meat sectors (beef, pork, chicken and lamb), albeit at different levels of investigation into system-wide uncertainties. Frequently cited examples of works on the poultry and sheep sectors include Pelletier (2008) and Biswas et al. (2010), respectively. For the pig sector, the author of this thesis has previously carried out an extensive literature review (McAuliffe et al., 2016) outside the current research.

For the beef sector, de Vries et al. (2015) summarised a wide range of LCA studies from around the world. It is noteworthy, however, that the popularity of beef LCA research has grown exponentially over the last three years. According to *Scopus*, there were 14, 26 and 24 papers published, respectively, in 2014, 2015 and 2016 under the search criteria “life cycle assessment” and “beef”; in 2017, this number jumped to 45. This growth in interest is likely a consequence of reports concluding that beef production, and particularly grazing systems, are extremely heavy contributors to global GHG emissions when evaluated on a mass-based FU (Springmann et al., 2016). Motivated by such a rapid increase in attention, the remainder of this subsection gives an overview of beef LCAs that have been published after de Vries et al. (2015). The selection of papers was conducted in March 2018 based on a *Scopus* search with keywords listed above and publication years of 2015 – 2018. The first 100 studies returned were then filtered according to the following rules: (1) written in English language; (2) primarily focuses on beef systems (rather than dairy systems or other species); and (3) is not primarily designed for end-point modelling. These criteria resulted in 14 papers.

In Brazil, Dick et al. (2015) conducted an LCA of beef cattle in two grassland systems. The first system was based on traditional grazing practices where animals can wander freely and receive little or no supplementation. The second system, termed “improved”, involved weekly rotational grazing and the introduction of winter forage species. The system boundary was from raw material extraction to farm gate and the FU was 1 kg liveweight gain (LWG). Data on beef production within the two systems were sourced from published literature, and GHG emissions were calculated according to IPCC guidelines. GWP for the traditional system was found to be 22.52 kg CO_2-eq per kg LWG, while GWP for the improved system was 9.16 kg CO_2-eq per kg LWG. This dramatic reduction in GWP was attributed to higher quality forage with increased digestibility in the alternative system, resulting in faster weight gains.

Mogensen et al. (2015) carried out a carbon footprint (CF) of beef production systems in Denmark and Sweden. The system boundary was from cradle to farm gate and the FU was 1 kg carcase weight (CW). Five Danish and four Swedish beef farming scenarios were developed, which were categorised depending on intensive or extensive production, and dairy or beef bred cattle. For feed production (pasture, silage and concentrates), C sequestration was considered based on IPCC guidelines and published literature. Grass-clover swards were included as part of an arable rotation where the swards remained for two to three years in a five-year rotation. Greenhouse gas emissions were estimated using a combination of IPCC values, e.g. for direct nitrous oxide (N_2O) and indirect N_2O via NH_3 and NO_3^-, and published book values for Nordic conditions (e.g. methane (CH_4) from enteric fermentation and manure management). The resultant CF ranged from 8.9 to 17 kg CO_2-eq per kg CW for the dairy-bull fattening systems, while the CF for cow-calf systems ranged from 23.1 to 29.7 kg CO_2-eq per kg CW. Carbon sequestration resulted in GWP mitigation across all scenarios; amongst them, CO_2 reduction was largest in the grass-based systems, although these systems still generated the highest CF values despite elevated C uptake.

Wiedemann et al. (2015) utilised LCA to examine the environmental impact of Australian beef and lamb being exported to the US. The system boundary was from cradle (in Australia) to the distribution warehouse in the US, and the FU was 1 kg retail ready meat. For beef systems, the study considered beef cattle bred in rangelands and finished on pasture, and dairy steers finished on grain feedlots for either 115 days or, for specialised breeds such as Wagyu, 330 days. Farm level data were obtained from governmental surveys and published case studies. Regionally tailored herd models were used to calculate feed intake and for predicting GHG emissions. Data on slaughtering and processing (such as cutting and chilling) were derived from an industry survey of meat processing plants in Australia. GWP ranged from 23.4 to 27.2 kg CO_2-eq per kg beef, with the grass-finished cattle performing least favourably. Across the three scenarios, the farming phase generated the highest GWP (93%), meat processing accounted for 4%, transportation 3%, while the warehousing had negligible impacts. However, the authors also considered differences in human edible protein conversion efficiency. Under this FU, pasture-based beef production performed considerably better than grain-fed beef by converting more non-human-edible protein into human-edible protein.

In an effort to capture temporal variations in on-farm GHG emissions, Hyland et al. (2016) assessed the CF of 15 livestock enterprises over two time periods three years apart (2009/10) and 2012/13). In addition to calculating farm-level emissions intensities, the authors also used a range of sensitivity analyses to investigate potential mitigation strategies. The system boundary of the study was set as cradle to farm gate, and the FU as 1 kg LW. Across the 15 livestock enterprises examined, five specialised in lamb, four specialised in beef, while six were mixed beef and sheep farms. Emissions intensities were calculated according to IPCC (2006) tier 1 and 2 guidelines. In tackling the issue of mixed farming allocation, where possible Hyland et al. (2016) used system expansion; however, in certain cases this was not possible due to a lack of differentiation and economic allocation was used instead. Between two data periods, lamb emissions were found to increase by 12%, while beef emissions decreased by 12%. However, these differences were not found to be statistically significant. Unsurprisingly, CH_4 emissions primarily resulting from enteric fermentation were the greatest GHG burdens across all enterprises. Regarding the scenario analysis aimed at reducing on-farm emissions, the authors suggested that the primary focus for farmers should be on improved resource use efficiency. The inclusion of legumes such as red (*Trifolium pratense*) and white (*Trifolium repens*) clover on suitable soils was also highlighted as an important technique to reduce fertiliser requirements.

Examining the impacts of Canadian grazing management strategies on GHG intensities from beef herds, Alemu et al. (2017) modelled a typical herd structure of 120 cows, four bulls and their progeny over an eight-year period. A range of different grazing strategies were considered: light continuous grazing for all cattle; heavy continuous grazing for all cattle; light continuous grazing for cow-calf pairs and moderate rotational grazing for backgrounded cattle; and heavy continuous grazing for cow-calf pairs and moderate rotational grazing for backgrounded cattle. The system boundary was set as cradle-to-farmgate, and results based on two FUs (LW and CW) were reported side-by-side. The authors used Holos (a Canadian whole-farm model) to estimate farm-level emissions, and soil carbon changes were considered using the Introductory Carbon Balance Model, while farm management data were sourced from previous management studies. Emissions intensities were found to have narrow ranges (14.5 – 16.0 kg CO_2-eq/kg LW; 24.1 – 26.6 kg CO_2-eq/kg CW) across the grazing scenarios; however, GHG emissions tended to decrease as stocking density increased. Inclusion of soil as a carbon sink reduced impacts by up to 25%. The authors highlight the complexities in crediting a grassland system as a carbon sink due to the extremely dynamic nature of carbon flows.

Berton et al. (2017) applied the LCA method to examine the environmental footprint of the integrated French-Italian beef production system. The system boundary was set as cradle to farmgate; however, unlike many other studies, this boundary accounted for a cow-calf operation in one country, France, with animals ready for fattening transported to another country, Italy. All inputs and outputs (including transportation) associated with each stage were accounted for, and impacts were scaled to a FU of 1 kg LW (described as bodyweight). The authors considered a range of impact categories made up of GWP, AP, EP, cumulative energy demand (CED) and land use (LU); reported as land occupation). Regarding allocation of burdens to coproducts of the cow-calf operation, a mass approach was adopted along with a sensitivity analysis to consider the effect of this assumption. Forty French farms and 14 Italian farms were modelled based on best available data. The authors found that the burdens arising directly from the farms were greater than upstream processes in general; the only exception to this finding was CED, where energy demand was higher for off-farm processes for production of feed and agrochemicals. In terms of total impacts, the authors highlighted positive correlations between direct environmental burdens (GWP, AP and EP) and resource requirements (CED and LU) and pointed out that agricultural policy design needs to account for multiple indicators rather than focusing on one.

In Italy once again, Buratti et al. (2017) compared the CF of conventional and organic beef production systems. Data were collected from two case study farms in the Umbria region of Italy, both of which operated as cow-calf systems rather than specialist fattening operations. The system boundary was from cradle to farmgate, and the FU was 1 kg LW of heifers and bullocks ready for slaughtering. Feed production primarily occurred on each of the farms, and burdens arising from fodder were modelled based on production data provided by the farms. The few imported products were treated as background processes and sourced from *ecoinvent* V3. Fertilising strategies differed between the enterprises. For example, the ‘organic’ system solely used livestock manure to fertilise feed crops, while the ‘conventional’ system used mineral N in addition to manure. Both systems transported excess manure to nearby but external cropland. GHG emissions were estimated using IPCC (2006) tier 2 guidelines for all foreground sources, and, regarding enteric fermentation, the authors estimated Y_m values (CH_4 conversion factors) according to the DE% (digestible energy) of the feed. The authors reported that lower GHG emissions were generated when producing organic feed due largely to lower mineral N requirements; however, interestingly, this did not translate to total CF rankings. The conventional system had a lower CF than the organic system, primarily driven by the shorter finishing times required.

de Figueiredo et al. (2017) examined the GHG balance and CF of three pasture-based beef finishing systems in Brazil. Three pasture systems all consisting of *Brachiaria* were defined as: a degraded pasture receiving no external inputs; a managed pasture receiving annual fertiliser with animals receiving strategic supplementation consisting maize bran (82%), milled soybean (14%), urea (3%) and mineral salt (1%) for a six-month period during dry season at a rate of 4 g/kg bodyweight; and a crop-livestock-forest integration system, a more complex system involving afforestation and rotational crop production and the same supplementation described under managed pasture. Both values were calculated using IPCC (2006) guidelines; the GHG balance was reported in terms of land area (1 ha), whereas the CF was reported as 1 kg LW leaving the farmgate. On an area basis, degraded pasture was found to have the lowest GHG balance, due to considerably lower stocking rates and no fertiliser requirement. Nevertheless, this finding was reversed in terms of 1 kg LW and degraded pasture was found to be the least efficient system due to low animal productivity. Between the two improved pastures, managed pasture was found to have considerably lower emissions (in terms of LW) than crop-livestock-forest, with livestock productivity again being a key factor. The crop-livestock-forest system brings its own merits in terms of other impact categories not assessed, such as improved biodiversity and utilising land to produce timber and crops as coproducts from the system. Overall, the authors conclude that land designated as degraded pasture should be improved wherever feasible. This study further questions the use of area as a FU for system-level environmental evaluation.

Florindo et al. (2017) used LCA methodology in combination with life cycle costing (LCC) to evaluate both the CF and economic performance of beef cattle in the Brazilian Midwest. The authors point out that LCA studies often recommend mitigation strategies to reduce environmental footprints while failing to account for economic viability, a trade-off they explicitly consider. Primary data for the study, including machinery costs and management activity, were collected directly from a beef farm comprising 1,350 ha of grassland. The farm maintains 1830 animals consisting of breeding stock as well as growing and finishing cattle. As part of the diversification strategy, the farm is split into four different production systems, differentiated by feeding regimes, stocking densities and slaughter weights. Feeding regimes were determined as with or without strategic supplementation which varied depending on the life stage of the cattle (e.g. creep feed). The protein mineral supplement included cornmeal (36%), soybean meal (12%) and urea (11%). Creep feed was made up of 30% cornmeal and 51% soybean meal, while a 14% protein ration provided based on LW consisted of 72% cornmeal and 18% soybean meal. GHG emissions were calculated according to IPCC (2006) tier 2 guidelines. Regarding LCC, the production system with the longest duration in terms of grazing was found to be the most cost-effective feed source, due to reduced supplementary feeding requirements. However, despite this positive aspect, it also resulted in the largest total financial cost due to lower stocking densities, and therefore, greater capital expenditure for land-use. The same finding was true for GHG emissions; higher stocking rates and lower grazing durations generated lower CFs, despite the subsequent lower finishing weights. This demonstrates the benefits of strategic supplementation, particularly in geographical regions affected by severe weather (extremely dry seasons in this instance).

Utilising interdisciplinary skills and expertise, Hessle et al. (2017) examined how Swedish beef and milk production systems could be environmentally and economically optimised under a range of different scenarios. Input was provided by experts in economics, LCA and supply chain management. The focus of this study was the environmental comparison of the reference situation (business-as-usual) with three hypothetical yet realistic scenarios. The three expertly-designed scenarios were based around Swedish environmental objectives and set as follows: an ‘ecosystem’ scenario aimed at reducing impacts on biodiversity; a ‘nutrient’ scenario which focused on optimising plant nutrient use and supply; and a ‘climate’ scenario primarily concerned with reducing anthropogenic GHG impacts. The overarching goal of each alternative scenario was to maintain or improve production efficiency, while simultaneously mitigating environmental impacts. Once the study panel had agreed upon the alternative systems, LCA models were constructed using a combination of literature and expert opinion. In most instances, the improved systems demonstrated reduced negative impacts. However, there were notable trade-offs; for example, the ecosystem scenario required more land being used as grassland to improve biodiversity, which in turn caused negative impacts on eutrophication (freshwater and marine) and cumulative energy demand across both beef and dairy systems. Despite this, the authors concluded that a common denominator in improving these livestock systems was more efficient use of resources such as energy and feed.

Tichenor et al. (2017) analysed differences in environmental performances between intensively managed grass-fed beef production and confinement dairy beef production systems in the Northeast of the USA. The system boundary was from cradle to farmgate and the authors considered hot carcase weight as the FU to maximise comparative potential with previous US studies. The impact categories considered were GWP, AP, EP, fossil fuel demand, water depletion and LU. For dairy beef, the authors adopted biophysical allocation at the ratio of 9.4:0.4:90.2 for beef/veal/milk, respectively.

They also considered economic allocation in a sensitivity analysis, at the rate of 7.8:0.9:91.3. Across GWP, EP, AP and LU, grass fed was found to have higher burdens than dairy beef. On the other hand, dairy beef required more fossil fuel and water than grass fed. The authors also considered impacts on a per ha basis, which resulted in lower AP and EP burdens for grass fed. A sensitivity analysis to account for carbon sinks in grassland was also considered. While this inclusion substantially reduced the GWP of grass fed, it was not enough to offset the benefits of productivity from DB. The authors echoed the argument of Berton et al. (2017) that future research should consider multifaceted aspects of grass fed systems that are socially important.

Wiedemann et al. (2017) examined resource use and GHG emissions associated with seven Australian feedlot beef systems. The authors adopted a gate to gate approach, with a primary focus on impacts arising from the grain-finishing stage. The FU for comparisons between the finishing stages was 1 kg LWG, while values for the entire system (including cow-calf enterprise) were reported as 1 kg LW. Three classes of cattle were considered: short-fed (55 – 80 days) for domestic market; mid-fed (108 – 164 days) and long-fed (> 300 days) for alternative export markets. Similarly, to Hyland et al. (2016), Wiedemann et al. (2017) found that CH_4 emissions aggregated across enteric fermentation and manure management were the most significant contributors to emissions intensities. Across the three management strategies, long-fed generated more GHG emissions than mid-fed which in turn generated more emissions than short-fed, due largely to the length of production cycles. The same rankings were observed for fossil energy demand (as MJ). However, the opposite rankings were noted for water consumption, an impact category with high importance in the arid regions of Australia. While the differences were not significant between short- and mid-fed, long-fed cattle had considerably lower freshwater usage due to reduced irrigated water usage. In terms of cradle to gate analysis, the finishing systems were found to contribute 26 – 44% of the total emissions intensity, with higher maximum impacts (up to 72%) recorded for total energy demand. The authors note that switches from pasture based to grain-based systems have reduced Australia's national emissions intensity from beef cattle, but these switches have been met with a trade-off of increased national energy demand. This signifies the complexities of drawing conclusions across multiple impact categories.

Willers et al. (2017) sought to identify environmental hotspots in semi-intensive beef production systems in Brazil's Northeast. The study accounted for two farms: the cow-calf operation and a separate but nearby finishing system. Similarly, to most beef LCA studies, the authors adopted a cradle to farmgate system boundary and a FU of 1 kg LW leaving the finishing farm. Primary data were gathered from the managers of both farms, while background processes were sourced from *ecoinvent* V2. The authors considered five impact categories: GWP (reported as climate change); AP (reported as terrestrial acidification); EP (reported as freshwater eutrophication), LU (reported as agricultural land occupation) and fossil depletion (FD). Following Berton et al. (2017), Willers et al. (2017) used mass allocation to disentangle burdens arising from coproducts at the cow-calf stage. Regarding the identification of hotspots, the authors diverted from conventional approaches and considered pasture processes as separate entities to their modelled livestock. This resulted in an unusual attribution of the overall burdens, whereby "grassland production" has higher effects on all impact categories than "livestock burdens", making inter-study comparison of the results (de Vries et al., 2015) rather difficult.

Bragaglio et al. (2018) analysed the environmental footprints of a range of different beef production systems in Italy utilising data collected from 25 farms. The systems studied were: specialised extensive; high grain fattening; intensive cow-calf constantly kept in confinement; and native breed (Podolian) maintained on pasture and finished in housing. The authors considered GWP, water depletion, LU, AP and EP within a system boundary set as cradle to farmgate and a FU of 1 kg LW. In terms of GWP, the intensive systems (high grain fattening and cow-calf confinement) were found to have lower impacts due largely to improved growth rates. However, the authors found that the systems with durations of pasture grazing (specialised extensive and Podolian) had lower AP than cow-calf confinement. There was no significant difference noted for water depletion, while high grain fattening and Podolian demonstrated the lowest burdens in terms of water quality (EP). Significantly higher LU was required for specialised extensive and Podolian; however, the authors also acknowledged that competition with human edible feed was lower for the grazing systems, particularly Podolian. A theme recurrent throughout grazing livestock LCA studies, namely the omission of ecosystem services and other societal benefits (e.g. improved animal welfare and meat quality) provided by grassland systems, is also highlighted by the authors. Bragaglio et al. (2018) conclude by acknowledging the importance of future LCA studies addressing these aspects of livestock systems that are more difficult to quantify.

Table 1.3 summarises the analytical approaches adopted by the 14 papers reviewed above, with a particular attention to their treatment of uncertainty. Overall, it demonstrates a considerable gap in knowledge within the livestock LCA community of uncertainty inherent in various farming systems, a key issue initially raised in Section 1.1. For example, none of the 14 studies used individual livestock data, which means intra-herd distributions of animal properties and performances could not be considered. Eight out of 14 papers did use farm-level aggregated data; however, only one of these studies included primary information on forage quality, a parameter widely known to be affected by farm management and, in turn, contribute to the uncertainty surrounding CH_4 emissions by enteric fermentation. Furthermore, none of the studies adopted site-specific emission factors for calculating GHG emissions. This is unsurprising given the timeframe required to measure GHG fluxes; yet, this also opens an opportunity to improve accuracy of on-farm emissions intensity at relatively low cost. Finally, only three studies conducted MC analysis, reiterating the lack of attention bestowed upon uncertainty on the whole (Imbeault-Tétreault et al., 2013). All papers reported uncertainty complexity Tier 1, defined by Igos et al. (2018) as the inclusion of sensitivity and scenario analyses. The three papers identified in **Table 1.4** which included MC analysis did not comply with the prerequisites for Tier 3, or advanced, uncertainty complexity, defined as using statistical analyses, such as correlation coefficients, to explain acknowledged uncertainties.

1.4 The North Wyke Farm Platform

As already mentioned in Section 1.1, studies carried out in this thesis, primarily Chapters 3, 4 and 5, will make extensive use of high-resolution data collected at the NWFP so that the degree of uncertainty surrounding LCA models can be quantitatively evaluated. While information specific to each analysis will be provided within the respective chapter, it would be useful to summarise here the overall farm design that is relevant across multiple chapters. Given the NWFP's role as a UK National Capability facility, this design was largely beyond the author's control.

The NWFP is located in Devon, a southwest county of England, UK (50°46'10"N, 3°54'05"W) and consists of three hydrologically isolated small-scale (21 ha) pasture-based livestock farms locally known as "farmlets" **(Figure 1.3)**. Each of the three farmlets is composed of seven fields as well as a winter housing facility and operates under a different pasture management system, with swards of: (1) permanent pasture (PP), which has not been reseeded for at least 20 years; (2) white clover (*Trifolium repens* cv. Aberherald)/high sugar perennial ryegrass (*Lolium perenne* cv. AberMagic) mix (WC), which aims to maintain 30% ground cover by white clover; and (3) high sugar perennial ryegrass monoculture (HS), which utilises the latest improved grass varieties under a regular reseeding programme. PP can be seen as business as usual, or a control, as approximately 82% of English grasslands are managed as permanent pastures; the remaining 18% are typically reseeded every 3-5 years (DEFRA, 2017). Of the latter 18%, 74% of livestock farmers sow at least part of their land with a clover mix (i.e. WC), while 62% of farmers include high sugar grasses in their swards (DEFRA, 2018). When successfully established, clover is generally considered to be beneficial both economically and environmentally because its nitrogen (N)-fixing properties reduce inorganic N requirements (Andrae, 2016). Grasses with higher levels of water soluble carbohydrate, on the other hand, are thought to reduce N losses to urine by improving N (or protein) use efficiency in the rumen while simultaneously improving production performance (Parsons et al., 2011).

The NWFP's cattle enterprises act as a finishing operation. Every autumn, 30 Charolais × Hereford-Friesian spring-born calves enter each farmlet at the point of weaning. At this time, animals are blocked between sexes and then randomly allocated to the farmlets from an adjacent but separate cow-calf enterprise, of which grasslands are permanent pasture similar to the PP system. After entering the NWFP, animals are typically housed from October to April to avoid destruction of soil structure during the wet season, then moved and kept outdoors on their respective pastures until they reach target weights of ca. 555 kg for heifers and 620 kg for steers and estimated meat quality scores (RPA, 2011) of "R" (conformation) and "4L" (fat). If animals do not meet these finishing criteria at pasture, a second housing period may be required. Throughout housing periods, animals are fed silage comprising grasses and legumes harvested from their own allocated systems (PP, WC or HS). While the NWFP's general principle is to finish cattle solely off pasture and silage, depending on the quantity and quality of silage produced in any year, strategic supplementary feed to balance energy and protein demands may be used and recorded. When strategic feeding occurs, its quantity is set at a uniform rate across animals within each farmlet. Cattle are housed in barns deep-bedded with barley (*Hordeum vulgare*) straw, and farmyard manure (FYM) produced is stored temporarily in middens until spreading in the next spring following first silage cut. Animals on each farmlet are rotated across the seven fields, which are also used for sheep grazing and silage production. Cattle and sheep never occupy the same field at the same time.

Data collection on the NWFP began in 2011, when it was established with the support of the Biotechnology and Biological Sciences Research Council (BBSRC) as a national capability. Prior to 2013, all three farmlets were composed of permanent pastures largely (>60%) dominated by perennial ryegrass. Between 2013 and 2015, the WC and HS farmlets were reseeded with white clover and high sugar perennial ryegrass, with the choice of cultivars based on the recommendation list of latest germplasms according to the National Institute of Agricultural Botany (NIAB). Throughout this transition period, the WC and HS fields underwent ploughing, ring rolling, harrowing, herbicide spraying, drill seeding and flat rolling; the PP system remained unaltered **(Table 1.4)**. On crop establishment, the HS (and PP) pastures started receiving standard N, P and K fertilisation and FYM, whereas the WC fields received a significantly lower amount of N, predominately in the form of FYM. Soil tests were conducted to assess nutrient status and health post-ploughing. The WC and HS soils were found to be generally acidic, resulting in a one-off application of lime to neutralise the acidity at variable rates between 150 kg/ha and 725 kg/ha. In addition, the WC system was found to be low in P levels, and consequently required higher levels of P_2O_5 application than the other two systems in the first year. Further information on the NWFP's history, soils and hydrology is provided elsewhere (Orr et al., 2016).

1.5 Thesis outline and structure

The remainder of the thesis will be split into five chapters, each with a distinct objective as detailed below. Collectively, they are designed to address the overall aim of the thesis: to reduce uncertainty associated with environmental foot-printing of meat production systems.

Chapter 2:

Objective 1: Quantify the environmental performance of livestock production systems using steady-state herd data that can act as a baseline for methodological comparison.

This first exercise is carried out using national-level datasets for pig production. Three impact categories (GWP, AP and EP) are considered under three production efficiencies (average, top 25% and top 10%). Implicitly, this approach assesses the environmental footprint of a "representative animal", whereby all animals within a given herd, or within a whole country in some cases, are assumed to perform equally. The results from this investigation form the comparative basis for subsequent methodological developments within this thesis and will also feed into the monogastric–ruminant comparison carried out as part of the final objective (Chapter 6). Uncertainty assessment in Chapter 2 aligns with complexity Tier 1 (basic) and Tier 2 (intermediate) as described by Igos et al. (2018).

Chapter 3:

Objective 2: Develop a novel approach to calculate carbon footprints of individual animals that constitute a herd.

This chapter calculates gate-to-gate (partial) CFs of all finishing cattle on the NWFP in 2015/2016. Pasture and silage quality are analysed to improve the accuracy of animal-originated GHG emissions, and in particular CH_4 emissions arising from enteric fermentation. The primary novelty of this study lies in the discovery that poorly performing animals with slower growth rates generate exponentially, rather than linearly, higher levels of greenhouse gas emissions. This suggests that representative animal approaches commonplace in the literature (and reproduced in Objective 1) may, in fact, be underestimating environmental burdens by failing to account for uncertainty caused by these extreme livestock. At the same time, the study also demonstrates the potential to drastically reduce GWP from cattle production through selection of environmentally friendlier, more productive animals. Consideration of uncertainty in Chapter 3 would be classified as complexity Tier 3 (advanced) according to Igos et al. (2018) due to the inclusion of correlation analysis i.e. between average daily gain (ADG) and GWP. Tiers 1 (sensitivity and scenario analyses) and 2 (MC analysis) are also addressed.

Chapter 4:

Objective 3: Derive site-specific emission factors for GHG emissions from soils through a static chamber field experiment.

Following the literature review in Section 1.3, it became clear that livestock LCA studies heavily relied on IPCC emission factors, which may or may not be applicable to specific sites of each study. In order to reduce this uncertainty, this chapter develops site-specific emission factors for the NWFP through a field experiment using static chambers. Plots are established on each of the three farmlets and received treatments of cattle urine and dung. Following a six-month sampling campaign, emission factors are developed for the subsequent use in LCA.

Chapter 5:

Objective 4: Evaluate the impacts of utilising site-specific emission factors on farm-level uncertainty in carbon footprint.

Combining the animal-by-animal method developed in Chapter 3 and emission factors derived in Chapter 4, the penultimate chapter provides a full CF model of beef production at the NWFP over two production cycles from 2014 – 2017, including the suckler-herd that supplies calves from outside the NWFP boundary. In addition to the N_2O emission factors based on original measurements, site-specific CH_4 conversion factors are also derived using a dataset from an experiment external to this thesis. Using on-site emission factors is shown to affect overall footprints, resulting in this instance with increases to the CF point estimate. Furthermore, the confidence interval for GWP is found to be narrower under on-site emission factors. Based on these findings, potential benefits of local GHG measurements are discussed. As in Chapter 3, uncertainty assessment in Chapter 5 covers all three Tiers defined by Igos et al. (2018).

Chapter 6:

Objective 5: Propose new LCA metrics (functional units) that can represent human nutritional value of different livestock production systems without end-point modelling.

Similar to the vast majority of agri-food LCA studies, Chapters 2, 3 and 5 all employ LCA based on a mass-output of a livestock production system. One of the major criticisms against this approach is that it can only account for product quantity rather than quality. To demonstrate that meat produced under different management strategies can have a profound effect on product quality, the final chapter looks ahead to potential approaches for more accurate comparisons of livestock systems within the LCA framework. Linking findings from Objective 1 (pork) and Objectives 2, 3 and 4 (beef), the study considers how relative CFs of beef, chicken, lamb and pork production systems are altered when the FU is changed to account for levels of omega-3 polyunsaturated fatty acids contained in the final products, and, more generically, the nutrient density of meat produced under different farming systems. The chapter concludes with a discussion on the implications and necessities of including product quality in comparative livestock LCA, and also summarises the strengths and limitations of the thesis in its entirety.

Table 1.1. Severity factors for selected chemicals associated with each impact category (Heijungs et al., 1997).

Impact category	Substance	Factor
GWP (CO_2-eq)	Methane (CH_4)	25
	Nitrous oxide (N_2O)	298
AP (SO_2-eq)	Ammonia (NH_3)	1.6
	Nitrogen oxides (NO_X)	0.5
	Nitrogen dioxide (NO_2)	0.5
EP (PO_4^{3-}-eq)	Ammonia (NH_3)	0.35
	Chemical Oxygen Demand (COD)	0.022
	Nitrate (NO_3)	0.1
	Nitrous oxide (N_2O)	0.27
	Nitrogen oxides (NO_X)	0.13
	Phosphorus (P)	3.06

Note: IPCC (2013) have since updated GWP factors. CH_4 and N_2O now equate to 28 and 265 CO_2-eq, respectively. These updated values are used in Chapters 3 and 5 while the older values adopted by the CML impact assessment (encompassing GWP, AP and EP) are used in Chapter 2.

Table 1.2. Overview of tests for validating data and uncertainty (Bauman and Tillman, 2004).

Type of test	Purpose of test
Completeness check	Check for data gaps in inventory or completeness of inventory
Consistency check	Check appropriateness of life cycle modelling and methodological choices given the defined goal and scope
Uncertainty analysis	Check the effect of uncertain data
Sensitivity analysis	Identify and check the effect of critical data
Variation analysis	Check the effect of alternative scenarios and life cycle models
Data quality assessment	Assess the degree of data gaps, approximate data and appropriate data

Table 1.3. A breakdown of various topics covered in the reviewed literature. Although some studies examine a range of impact categories, emission factors pertain to GWP only.

Study	LW[a] change (over time)	No. of animal categories[b]	Farm data source	Feed quality	Emission factors	Monte Carlo analysis
Dick et al. (2015)	ADG[c] split between first year and subsequent years	9	National statistics	Existing literature	IPCC Tier 2	×
Mogensen et al. (2015)	Fixed ADG per system	6	Existing literature	Existing literature	Danish specific & IPCC Tier 2	×
Wiedemann et al. (2015)	Fixed ADG	4 (beef only)	Farm	National statistics	Australian specific	✓
Hyland et al. (2016)	Monthly LWG for growing stock	Unspecified	Farm	National statistics	UK specific & IPCC Tiers 1 & 2	×
Alemu et al. (2017)	Fixed ADG per category	7	Existing literature and experimental data	Measured data	IPCC Tier 2 and Canadian specific	×
Berton et al. (2017)	ADG varied across three points of time on each farm	5	Farm	Existing literature	French specific and Tier 2 IPCC	×
Buratti et al. (2017)	Fixed ADG per animal category	7	Farm	Existing literature	IPCC Tier 2	×
de Figueiredo et al. (2017)	Fixed ADG per system	Unspecified	Existing literature	Unspecified	Brazilian specific and IPCC Tier 1	×
Florindo et al. (2017)	ADG varied by age and scenario	4	Farm	Unspecified	IPCC Tier 2	×
Hessle et al. (2017)	Unspecified	5	National statistics	Existing literature	IPCC (unspecified tier)	×
Tichenor et al. (2017)[d]	Unspecified	6	Existing literature	Not applicable[d]	Not applicable[d]	×
Wiedemann et al. (2017)	ADG varied by farm and scenario	3	Farm	National statistics	Australian specific	✓
Willers et al. (2017)	Unspecified	4	Farm	Unspecified	IPCC (unspecified tier)	✓
Bragaglio et al. (2018)	Fixed ADG per system	3	Farm	Existing literature	IPCC Tier 2	×

[a] Liveweight; [b] By age, sex and breed; [c] Average daily gain; [d] Only examines land use.

Table 1.4. Farm activities carried out under each system of the North Wyke Farm Platform.

Activity	PP	WC	HS
Ploughing	✗	✓	✓
Rolling	✓	✓	✓
Harrowing	✗	✓	✓
Seeding	✗	✓	✓
Fertiliser spreading	✓	✓	✓
Herbicide spraying	✗	✓	✓
FYM spreading (solid)	✓	✓	✓
Liming	✗	✓	✓
Mowing	✓	✓	✓
Silage making	✓	✓	✓

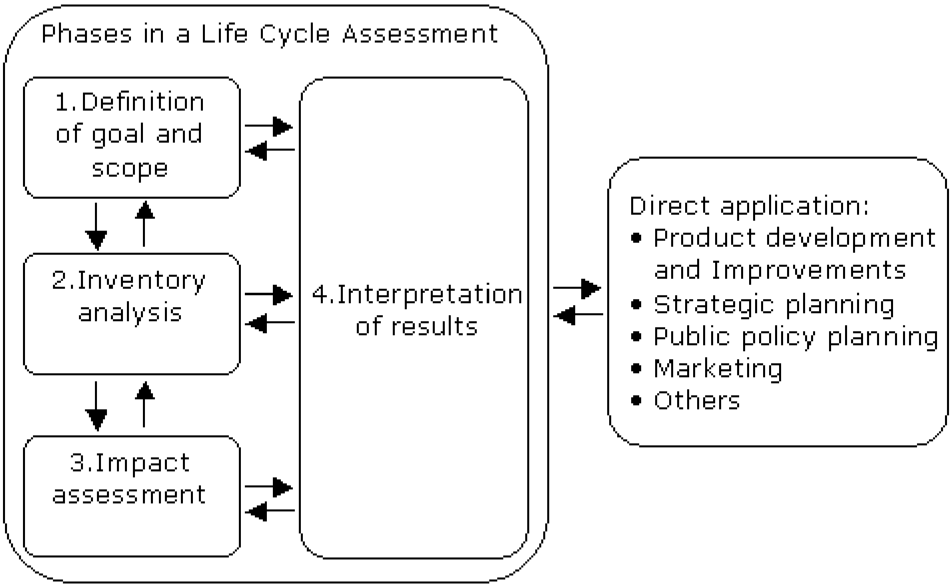


Figure 1.1. The systematic steps involved in carrying out an LCA study.

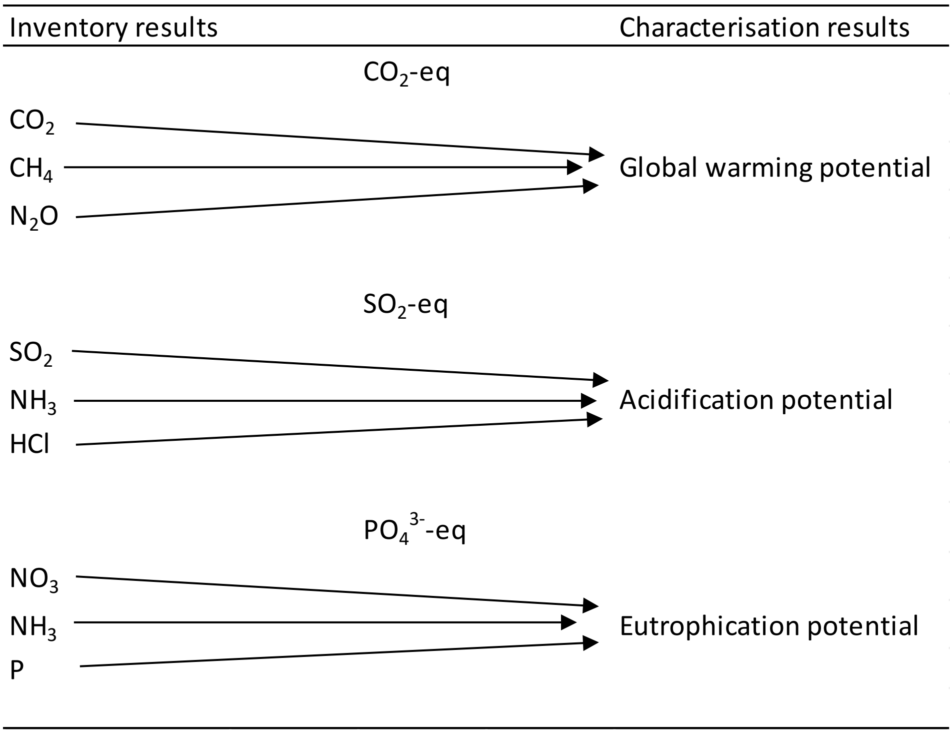


Figure 1.2. Emissions and losses characterised into appropriate impact categories. Sourced from Bauman and Tillman (2004).

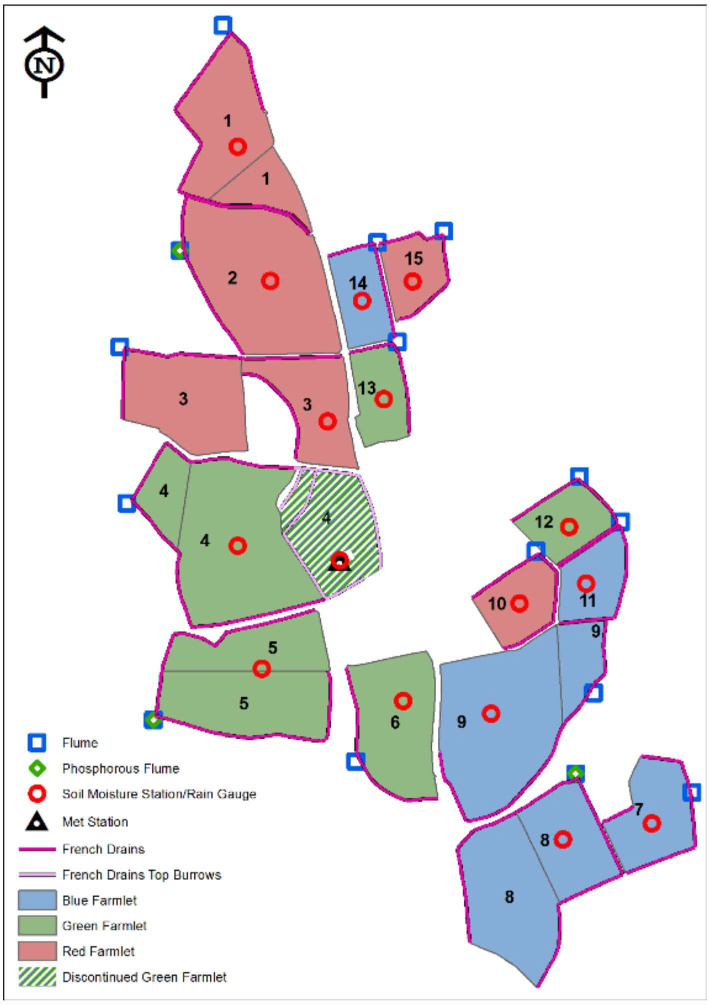


Figure 1.3. Map of the North Wyke Farm Platform in Devon, UK. Green = permanent pasture (PP); Blue = white clover/high sugar grass (WC); Red = high sugar grass monoculture (HS).

Chapter 2 - Environmental trade-offs of pig production systems under varied operational efficiencies

2.1 Introduction

As discussed in Chapter 1, ruminant animals are the primary drivers of agriculture-related global warming through enteric fermentation. Notwithstanding, recent evidence suggests that production of monogastric animals also require significant attention, as they compete for human edible-food and land resources—more so than pasture-based ruminant livestock, and also contribute to the environmental burden through water and atmospheric pollution (Wilkinson and Lee, 2018). In particular, pork is the most consumed meat globally (OECD, 2017), and its production is estimated to emit 668M tonnes CO_2-eq yr^{-1}, or 9% of total livestock emissions (Gerber et al., 2013). As the author of this thesis previously concluded following a comprehensive review of recent pig life cycle assessment (LCA) studies (McAuliffe et al., 2016), it is imperative to develop pig production systems, as part of a food security framework, that provide the right balance between economic, environmental and societal sustainability. To date, various authors have demonstrated that improved sow efficiency, through higher numbers of piglets born alive and reduced dry periods, can decrease environmental burdens (Reckmann and Krieter, 2015). Furthermore, higher feed conversion efficiency (FCE) has also been shown to reduce the environmental impact and land-use per pig unit, as emissions and losses associated with the feed production stage become smaller (Nguyen et al., 2011). However, research investigating these effects on the system-wide footprint is rather limited, and, consequently, the environmental benefit of economically improved pig operations is not clearly understood.

In order to fill this gap in literature and at the same time provide a baseline for methodological development carried out later in the thesis, this chapter applies a common LCA framework to national-level information of intensive pig production systems in the Republic of Ireland (RoI) and investigates their environmental performances under different production efficiencies. RoI was selected as a case exemplar primarily for three reasons: (1) the lack of previous LCA studies covering the pig industry; (2) the quality and detailedness of industry statistics; and (3) the proximity of production systems to those in the UK, where cattle production data were obtained for subsequent chapters. Pig production is the third most important agricultural sector in RoI based on gross agricultural output (Teagasc, 2016). Yet, contrary to the country's beef and dairy sectors that have previously been examined for their environmental impacts (Casey and Holden, 2005, Casey and Holden, 2006), and despite nationwide discussions on the merits of LCA in national GHG (greenhouse gas) evaluations (Schulte et al., 2011), the Irish pig industry has not been the subject of a systems study to date.

As of June 2015, there were 1.54 million pigs in RoI and, with an annual net production of just over 276,000 tonnes, the national self-sufficiency rate was 195%; nearly half of total production was exported. Similarly to many parts of the UK, most pig production in RoI occurs on large-scale integrated units, where piglets are born, weaned and fattened on the same farm. On these farms, feed is typically purchased from specialised production mills, but with the recent volatility of international cereal prices, a small number of Irish pig farmers have constructed their own on-farm mills to minimise costs and maximise nutritional control over their feed formulations. In addition to the baseline analysis whereby feed is assumed to be mass-produced, the present study investigates the effect of this 'local feed' movement on the environment footprint. While a range of LCA studies have considered differences in feed composition (Garcia-Launay et al., 2014, Ogino et al., 2013, Stone et al., 2012), no identified studies have considered the location and the ownership of feed mills.

In RoI, 7.4% of the total agricultural land is used for arable crop production and the country is close to self-sufficiency (encompassing human, animal and industrial uses) for major cereals (DAFM, 2009). However, many feed mills source a significant portion of cereal ingredients from overseas, especially when the international market is in a favourable condition (in regard to cereal prices and exchange rate). Replacing these cereals with domestically grown barley (*Hordeum vulgare*) and wheat (*Triticum* spp.) could potentially contribute to lower total transport distances, more efficient use of manure (nutrient balancing) and, perhaps to a lesser extent, long-term food security. The present study tests this hypothesis by investigating whether the reduced transportation, when coupled with domestic conditions for crop production (and the associated emissions), would alter the overall LCA results. Finally, four sets of sensitivity analyses and a MC analysis are conducted to evaluate the effects of uncertainties associated with model assumptions commonplace amongst LCAs using aggregate statistics. The sources of uncertainties considered are: different allocation methods, land use change (LUC), utilisation of pig manure by crop farmers, on-farm energy usage, and EFs for on-farm activities.

2.2 Materials and methods

In this study, LCA was applied to the Irish pig industry under both a typical industry setting (baseline analysis) and altered production systems (scenario analyses).

2.2.1 Goal, scope and functional unit

The primary goal of this study was to compare environmental performances of intensive pig production units operating at different efficiencies and evaluate the effectiveness of alternative strategies to improve sustainability of the industry. The system boundary for the baseline analysis was set as being from the 'cradle', or the production of input materials, to the end of the slaughtering process **(Figure 2.1)**. The environmental performance of each system was evaluated with the functional unit of 1 kg carcase weight (CW), of finishers and cull sows combined, as measured at the time when the intermediate product (dressed carcase) exits from the slaughterhouse. This functional unit was adopted to represent a wide range of pigmeat both (a) directly sold to retailers, as well as (b) initially distributed for secondary processing. Consequently, secondary processing and supply chain distribution beyond the abattoir were excluded from the model. The aforementioned review by the author of this thesis has noted, however, that cross-system comparison of environmental performances is extremely challenging when scopes and functional units are not shared between different analyses (McAuliffe et al., 2016). Motivated by this criticism, outputs based on the auxiliary functional unit of 1 kg liveweight (LW), as measured at the time when the intermediate product (live animal) exits from the farm gate, is also reported in this study. For the conversion of LW to CW, a kill-out rate of 76% was assumed based on data from 2014 (Teagasc, 2014).

2.2.2 Life cycle inventory analysis (LCI)

2.2.2.1 Feed production

Feed composition data for the baseline analysis were obtained from a large-scale commercial mill in RoI. These data were representative of commercially available feed rations used in the Irish pig industry during February 2015. Diet formulations were distinguished between dry sows, lactating sows, weaners and finishers, and replacement gilts that have reached the finishing weight but are yet to be served were assumed to consume the same amount of feed as dry sows. The major ingredients for these feed rations included barley, maize (*Zea mays*), soybean (*Glycine max*) products and wheat **(Table 2.1)**. All rations were formulated using the principle of least-cost rationing and balanced for macro and micro nutrients through the addition of supplements (mineral premix and synthetic amino acids: SAA) to meet animal requirements (maintenance and production). Environmental implications of using premix supplements were considered to be the same as the production of calcium carbonate (Mosnier et al., 2011). Environmental burdens of SAA, of which mass accounted for < 1% of total mass of ration, were excluded from the current LCA model due to unavailability of commercial sensitive data pertaining to the exact SAA composition within the recipe. A similar approach has been employed by Dourmad et al. (2014) and Nguyen et al. (2010b) and, while SAA have a large environmental footprint when evaluated on a per kg basis (Garcia-Launay et al., 2014; Mosnier et al., 2011), those arising from their production process at a system-level are generally small due to minimal quantities mixed into the feed (Strid Eriksson et al., 2005). No medicines or growth-promoting agents were included in the compound feed rations. The nutritional composition of the feed ingredients presented in **Table 2.2** was compiled based on data from (FAO, 2015a). Background data for crop production together with associated yields and environmental burdens were sourced from the Agri-footprint database (Durlinger et al., 2017), in which impacts of pesticide application events were considered, while those of upstream production were not. Based on data provided by the mill manager, it was assumed that 11 kWh of national grid electricity were used to produce 1000 kg of the mixed feed. As this feed was wet-mix, heat was not required for compression.

Information on origins and transportation of crop ingredients was provided by the mill and an importation company. As of February 2015, soybean products were imported from Argentina and shipped from Rosario Harbour. French maize, wheat and beet (*Beta vulgaris*) pulp were transported from Boulogne. Barley and wheat from the UK were delivered from Liverpool, while premix supplements were transported by road from Belfast. All sea-based cargo was delivered to Ringaskiddy harbour in Cork, RoI, and the nautical distances were calculated using Portworld (2016). From Ringaskiddy, these ingredients were transported using trucks, and the road-based distance for this segment was calculated using a geographical information system **(Table 2.2)**.

The environmental burdens arising from crops with multiple outputs were allocated by means of economic allocation. While splitting the responsibility of downstream emissions and losses into multiple upstream production processes could potentially disrupt mass and energy balances (Weidema and Schmidt, 2010), system expansion to cover the entire value chain of upstream products such as soybean oil and rapeseed (*Brassica napus*) meal was considered to be impractical given the scope of the present study (Ardente and Cellura, 2012). Following the recommendation by preceding studies that assignment of environmental burdens between crop co-products is best carried out by way of economic allocation (Williams et al., 2006), this method was adopted for background crop processes of the baseline analysis. Economic values of co-products were adopted from the Agri-Footprint database (Durlinger et al., 2017), of which primary data originate from Vellinga et al. (2013).

2.2.2.2 Pig production

Herd performance data were based on national statistics compiled by Teagasc (2014). These data covered 84,000 sows or 56% of the national breeding population. While farm size in the original record ranged from less than 100 sows to over 2,500 sows, the present study was carried out for the average herd size of 752 sows. Three sets of productivity data were used in this study (Teagasc, 2014): those representing farms with an average herd performance (AVG), the top 25% farms (T25) and the top 10% farms (T10), as measured by the number of pigs produced per sow and FCE of growing pigs. Consequently, three representative farms were set up for the baseline analysis **(Table 2.3)**.

Herd dynamics, including the schedule of replacement, was mathematically estimated for each of the three representative farms under the assumption that they are operating at steady state. Adult males were excluded from the inventory because of the disproportionally large number of sperm doses produced by a single boar under artificial insemination systems (Knox, 2016). The derived information shows that animals on the T10 herds tend to stay on farm for a longer period of time than the T25 herds, but meanwhile consume less than the T25 farms **(Table 2.3)**. The number of piglets born alive per sow was highest for T10, and this led to the higher sow feed intake, particularly at the farrowing stage (Teagasc, 2014). The T10 herds also had the lowest mortality rates across all stages of production. Carcase yields between the three categories were similar, suggesting that the difference in production efficiency is mostly attributable to better management of nutrition and health, rather than the difference in the target market. Based on local data provided by McCutcheon (2012), energy usage on farm was assumed to be 28 kWh per head (including both sows and finishers), of which 53% was consumed in the form of metered electricity and 47% in the form of processed light fuel oil used predominately for underfloor heating and ambient temperature regulation.

Pig manure in RoI is typically utilised as an organic fertiliser. On the majority of pig farms, animals are housed on slatted floors, where manure drains, assisted with water hosing, into an underground storage tank. Manure is usually stored in temporary tanks for less than one month, and then pumped out to an outside storage tank where it remains until receiving farmlands are ready for nitrogen (N) application. The pig units are typically large-scale indoor enterprises, and most pig farmers do not own enough land for arable production to spread the entire manure-output on (Nolan et al., 2012). Consequently, the manure is often transported to nearby arable farms for utilisation. In this study, it was assumed that manure was transported 10 km to receiving farmland. Diesel energy required for spreading manure was assumed to be 21 MJ per 1000 kg (Nguyen et al., 2010b, Reckmann et al., 2013), mostly attributable to the use of a tractor and manure spreader. Both the positive and negative effects of pig manure were considered in the baseline analysis, the former as a cause to reduce the demand for manufactured fertiliser and the latter as a source of ammonia (NH_3), methane (CH_4), nitrate (NO_3^-), nitrous oxide (N_2O) and phosphate (PO_4^{3-}) losses.

2.2.2.3 Slaughterhouse process

Most LCA studies that include the slaughterhouse within the system boundary demonstrate that, in comparison to primary production, the environmental impacts arising from this process are minor (Nguyen et al., 2011). Since primary data from Irish slaughterhouses were unavailable, data for the slaughtering process were taken from Reckmann et al. (2013), as their production environment in Germany was deemed most similar to the Irish situation. These authors report energy usage and emissions associated with the abattoir, while assuming that waste products and by-products are disposed of as biodegradable materials. Detailed inventory data prepared for the baseline analysis can be found in **Table 2.4**. Water use was not included due to the finding by Reckmann et al. (2013) that it had minimal impacts on global warming potential (GWP), acidification potential (AP) and eutrophication potential (EP). The carcase yield (kill-out %) for each representative farm, obtained from Teagasc (2014), is listed in **Table 2.3**.

2.2.2.4 Emissions and losses

Emission factors used in this study are provided in **Table 2.5**. The parameters for CH_4 emissions were taken from the Irish National Inventory Report (Duffy et al., 2017), while N_2O emissions were calculated using IPCC (2006) guidelines. NO_X and NH_3 emissions were calculated according to the methodology reported in Nguyen et al. (2011). Nutrient contents in manure were estimated using the nutrient balance, where the N, P and K contents in body tissues were subtracted from those in feed (Poulsen et al., 2001). Once applied to farmland under typical Irish conditions, 50% of manure N and 100% of manure P and K were assumed to become available for plant uptake (Government of Ireland, 2010). For P, 3% of this value is assumed to become lost through leaching (Nguyen et al., 2011). The potential impact of using site-specific EFs (rather than region-specific EFs) on uncertainties of LCA results will be investigated in Chapters 4 and 5.

The reduction in GHG emissions due to avoided production of manufactured fertiliser was estimated to be 6.6 kg CO_2-eq/kg fertiliser N (Wernet et al., 2016), 2.7 kg CO_2-eq/kg fertiliser P and 0.8 kg CO_2-eq/kg fertiliser K (Nielsen et al., 2007). Energy savings associated with reduced on-farm activities were assumed to be 0.4 MJ diesel/1000 kg fertiliser N (Nguyen et al., 2011). The associated reduction in emissions from soil was accounted for in **Table 2.5**. Reduced emissions from P and K application were not included in the model due to their small quantities, which were assumed to be spread together with N fertiliser. The complete LCI data for 1000 kg LW at the farm gate are given in **Table 2.6**.

2.2.3 Impact assessment and interpretation

SimaPro 8.1 (www.simapro.com) was used to model the studied systems. The three impact categories previously identified to be important for pig LCA studies (McAuliffe et al., 2016), namely GWP, AP and EP were estimated for the three representative farms with varying levels of productivity using the (CML, 2013) baseline impact assessment method. The outputs for the baseline analysis were expressed, respectively, in units of kg CO_2-eq kg CW^{-1}, g SO_2-eq kg CW^{-1} and g PO_4-eq kg CW^{-1}. Of the various sources of uncertainties surrounding LCA outputs, the effect of those inherent in livestock performance and farm management was assessed through: (a) the comparison of the three representative farms as discussed in Section 2.2.2, and (b) a range of scenario and sensitivity analyses as outlined below. Furthermore, the effect of uncertainties related to on-farm emissions was evaluated by means of Monte Carlo analysis and the resultant outputs were compared pairwise between the three representative farms. For the latter procedure, parameters were randomly drawn over 1000 iterations from the distributions summarised in **Table 2.7**.

2.2.3.1 Scenario analyses

For the first scenario analysis to examine the environmental implications of on-farm feed milling (see Section 2.1), data were collected from a small-scale farm-operated mill in the south of RoI. The data inventory presented on the right-hand side of **Table 2.1** replaced the baseline inventory on the left-hand side. Based on information provided by the mill manager, it was assumed that 30 kWh of electricity was used to process 1000 kg of feed, the level far above what was assumed for the large-scale specialist mill (11 kWh) in the baseline analysis. Since the mill is located adjacent to the piggery, the on-road transportation process linking the feed mill to the representative farms was eliminated from the model **(Table 2.6)**. For the second scenario analysis to examine the consequences of reduced transport distances, all imported cereals in the baseline inventory were replaced by domestically produced counterparts of the same quantity. To be consistent with the baseline analysis, data related to domestic crop production were also sourced from Agri-footprint (Durlinger et al., 2017).

2.2.3.2 Sensitivity analyses

The economic allocation method was used in the baseline analysis to separate environmental burdens associated with crops with more than a single material flow. A sensitivity analysis was conducted, here using mass-allocation, in order to test the robustness of the baseline results. This analysis was performed on all crops that had multiple outputs, for example, meal and oil produced from soybean and rapeseed.

Due to the relatively small scale of the Irish pig industry, the baseline analysis of the present study assumed that changes in feeding strategy on Irish farms would not cause LUC elsewhere in the world. Recent research has shown, however, that the inclusion of LUC in the assessment of soybean production systems can increase the resultant GHG emissions by as much as nine-fold when the entire crop-growing area is assumed, somewhat unrealistically, to have been forest previously (Maciel et al., 2016). Under a more reasonable assumption, a UK study by Audsley et al. (2009) posited that, when LUC is included in the model, up to 40% of the country's food-sector emissions would originate outside the country. Given the significance of such a potential impact, a sensitivity analysis to examine the potential effect of LUC was conducted using information compiled by Durlinger et al. (2017) in conjunction with PAS2050-1 (BSI, 2011). Emissions arising from LUC were estimated for rapeseed (Germany), soybean (Argentina) and wheat (RoI, Denmark and the UK). For production of barley (RoI and the UK), maize (France) and sugar beet (France), land transformation was deemed unnecessary (Durlinger et al., 2017).

In addition, several on-farm assumptions were deemed to require sensitivity analyses. First, the inclusion of the fertiliser offsetting effect in the baseline analysis (where manure N, P and K replace inorganic nutrients) implicitly assumes that pig manure is perfectly utilised by receiving farmers. Although pig manure is a useful by-product, it is difficult in reality to match demand and supply without wastage. Therefore, a sensitivity check was conducted to examine the effect of this offsetting on the overall results by assuming the other extreme case, whereby manure is applied to arable land in addition to manufactured fertilisers (i.e. in excess of crop nutrient requirements), resulting in no reduction in fertiliser production. Additionally, while the on-farm energy usage in this study was assumed to be 28 kWh per head, preceding studies show that this value ranges widely across pig farms in RoI. Therefore, using the upper limit (45 kWh per head) and lower limit (18 kWh per head) reported by McCutcheon (2012), two additional versions of models with high and low energy usage (retaining the electricity–fuel oil ratio of 53:47) were generated to examine the effects of this value on the overall environmental footprint.

2.3 Results and discussion

The environmental impact per kg CW obtained from the baseline analysis is displayed in **Table 2.8**. A detailed breakdown of contributions from all system processes is provided in the Appendix to the current chapter **(Tables A2.1-A2.3)**.

2.3.1 Global warming potential

GWP of the average (AVG) farm was estimated to be 3.5 kg CO_2-eq/kg CW, with the 95% confidence interval (accounting for uncertainties surrounding on-farm emissions) ranging between 3.1-3.9 kg CO_2-eq/kg CW. Based on the point estimate, the largest GWP hotspot was emissions arising from feed production, accounting for 58% of the total impact **(Table 2.8)** at a level comparable to other European studies (MacLeod et al., 2013, Parsons et al., 2011). Of feed-related impacts, the finisher diet accounted for 65%. Maize had higher emissions than other crops driven primarily by its mass input, wet-mill processing into maize bran and, to a lesser extent, more intensive fertiliser usage when compared to wheat and barley (Durlinger et al., 2017). Road and sea transport together accounted for 8% of total feed-related emissions. Transportation from Argentina by cargo ship generated 19% of the GWP attributable to soybean products, the only group of feed ingredients originating outside Europe. All other crop ingredients had considerably lower sea transportation impacts (< 2%).

On the farm, CH_4 emissions from manure management and enteric fermentation respectively generated 23% and 5% of total GWP, closely following the results reported by MacLeod et al. (2013). N_2O emissions arising from manure storage produced 3% of total GWP, while N_2O emissions from manure application produced 7%. The usage of national grid electricity accounted for 4% of total emissions, while light fuel oil burned in a non-condensing boiler was shown to have a relatively small effect (1%). Of emissions displaced in the arable sector, the reduction of N production resulted in a 9% saving of total emissions; on the other hand, the effect of replacing P and K fertiliser production was less profound (1%). Slaughtering accounted for 9% of total GWP kg CW^{-1}, of which electricity was responsible for 79%. This result is similar to the finding by Reckmann et al. (2013), who reported that 7% of total GWP was generated at the slaughterhouse. Contributions from other processes, including farm traction and transport of feed from mill to farm, were all comparatively minimal.

2.3.2 Acidification potential

AP for the AVG farm was estimated to be 43.8 (38.5-48.7) g SO2-eq/kg CW. NH_3 emissions from manure storage (indoor and outdoor combined) and application to crop fields respectively accounted for 26% and 28% of the total AP, making NH_3 losses the largest contributor to this impact category. Avoided NH_3 emissions from replaced inorganic fertiliser resulted in a 4% decrease from the level of AP that would otherwise have been produced, again insufficient to offset the large emissions arising from manure application. Environmental burdens resulting from NO_X were negligible (< 1%). Feed production accounted for 45% of the total AP, of which finisher feed represented 66%. These figures are comparable with Nguyen et al. (2011) where feed generated 36% of AP, while Reckmann et al. (2013) reported a slightly lower 23% contribution from feed. In the current study, maize (27%) and barley (26%) were the highest feed-related hotspots. Sea-based transportation accounted for 1.4% of total AP. The slaughterhouse generated 3% of the total AP, of which SO_2 emissions from combustion during electricity production accounted for 83%.

2.3.3 Eutrophication potential

EP for the AVG farm was estimated to be 32.1 (28.0-36.5) kg PO_4-eq/kg CW. Feed production was the highest contributor to EP, accounting for 51% of the total value. Similarly to AP, barley and maize were the primary sources, producing 28% and 22% of feed-related burdens, respectively. Losses of eutrophying substances such as NH_3, NO_3^- and PO_4^{3-} were the primary sources of EP from crop production. NH_3 emissions from farm management and manure spreading generated 17% of the total EP, while losses of NO_3^- from organic fertiliser application amounted to 19%. Environmental burdens associated with PO_4^{3-} from manure application on the receiving arable farms were low (1%). The slaughterhouse had a higher impact on EP than GWP and AP, totalling to 13%. The majority (85%) of these burdens stemmed from higher biochemical oxygen demand (BOD), chemical oxygen demand (COD), and increased losses of N and P to water.

2.3.4 Effect of herd performance

As discussed in Section 2.2.2, the three representative farms with different levels of productivity (AVG, T25 and T10) were differentiated by feed intake, mortality, growth rates and, to a lesser extent, carcase yields. **Table 2.8** indicates that improvements in production efficiency generally lead to smaller environmental footprints. Between the average (AVG) farm and the T10 farm, a 9% improvement in feed conversion ratio (from 2.49 to 2.27 kg/kg, as calculated from Table 3) is met by 6%, 12% and 15% decreases in GWP, EP and AP (p = 0.01, p = 0.03 and p < 0.01 based on Monte Carlo pairwise comparisons), respectively. It should be noted, however, that the present method used a fixed emission factor per head for CH_4 from manure production, which was not adjusted for reduced feed use per kg meat production. These percentages should therefore be seen as the lower limits, rather than the expected values, for the effect of improved farm productivity. Differences in GWP, EP and AP between the AVG and the T25 farms were also found to be systematic ($p < 0.01$, $p = 0.05$ and $p < 0.01$, respectively). Further discussions on approaches to account for individually different levels of manure-originated CH_4 emissions are given in Chapter 3.

The differences in environmental performances between the two improved herds were not as clear-cut (all $p > 0.10$). The T25 herd finished pigs in less time than T10 while the T10 herd consumed less feed in total **(Table 2.2)**, leading both their CH_4 emissions and the overall GWP to be comparable against one another. However, the T25 herd generated more N and caused larger losses of NH_3 and NO_3^- due to higher feed intake, and as a result larger AP and EP were predicted compared to T10. As a consequence, environmentally speaking, neither of the improved herds were strictly preferable over the other herd demonstrating the complexities of trade-off assessments. Economically speaking, lower costs associated with less feed consumption, together with increased throughput of liveweight generally led T10 to have higher profit margins, followed closely by the T25 herds (Teagasc, 2014). Based on the observation that the farms with higher levels of productivity (T25 and T10) generated lower environmental footprints than the AVG herd, it is plausible to conclude that improvements in animal performance metrics are more likely to be positively correlated with environmental sustainability. This finding is in agreement with Nguyen et al. (2011).

2.3.5 Scenario and sensitivity analyses

Figure 2.2 summarises main findings from the scenario and sensitivity analyses for the average (AVG) farm. Detailed results for all three herds (with different production efficiencies) are provided in **Table 2.9**. All values are reported as percentage change from the baseline results.

2.3.5.1 Scenario analyses

Some notable differences were observed as a result of replacing feed from the large-scale commercial mill with feed from the small-scale on-farm mill; GWP reduced by 13%, AP increased by 14% and EP increased by 6-7%. These differences between the commercial and on-farm feed mills were largely driven by the ingredients, rather than the milling method. For example, the lower GWP associated with the on-farm mill primarily resulted from lower maize bran usage (associated with 0.82-0.84 kg CO_2-eq per kg on a dry matter basis) compensated for by larger quantities of cereals (0.34-0.36 kg CO_2-eq for barley and 0.26-0.33 kg CO_2-eq for wheat, both on a dry matter basis), a combination that tends to generate a lower carbon footprint due to reduced energy requirements for wet milling of maize (Durlinger et al., 2017). Although more electricity was used per 1000 kg feed produced at the on-farm mill, this had little impact on the overall GWP (< 1%). Increases in AP and EP are explained by the larger quantities of soybean meal included in the diets, which has the highest CP content (51.8%) of all the ingredients. This resulted in larger quantities of N in manure, increasing potential losses of NH_3 and

NO_3^- **(Tables A2.4-A2.6)**. For example, NH_3 emissions (measured in g SO_2-eq) and NO_3^- losses (measured in g PO_4-eq) were both 33% higher using the on-farm mill diets. These findings alone warrant further research on economic-environmental trade-offs surrounding feeding strategies, as 'least-cost' ration formulations are solely driven by the market price of commodities and do not reflect differences in upstream processing requirements or indeed environmental costs attributable to different rations. More immediately, these conflicting results demonstrate the complex nature of interpreting LCA studies and disseminating results to key stakeholders (Guinée et al., 2011). On one hand, advising farmers to include more wheat and barley and less maize (particularly processed maize such as bran) seems to be a logical assessment as the present result suggests subsequent GWP decreases. However, as burdens generated from high protein crops such as soybean products produce higher levels of N in manure resulting in higher AP and EP (Garcia-Launay et al., 2014, Mosnier et al., 2011, Ogino et al., 2012), it is not immediately clear which option is environmentally more desirable when their FCE are comparable. This trade-off needs to be analysed in a local context, taking factors such as the current level of water quality into consideration, before recommendations are communicated with pig producers in the region. In any case, the composition of the ration is a direct consequence of the nutrient demands of the pig and the availability and price of feed ingredients in a particular region, and therefore any potential amendments in formulation are likely to be limited.

Despite lower distances travelled and higher yields achieved under Irish conditions, replacing imported wheat and barley with domestically-grown cereals had minimal effects (1%) across all impact categories. The slight increases observed to GWP and AP are mainly attributable to increased emissions, which was triggered by Irish farmers' general preference towards organic fertilisers (not limited to pig manure) compared to French and UK farmers (Durlinger et al., 2017). Marginally lower EP values occurred as less NO_3^- was leached on Irish crop farms, due to a higher retention rate of crop residues and less usage of inorganic fertilisers. The present finding that the replacement of imported cereals with domestic crops does not considerably alter the LCA results supports the view by (Dalgaard et al., 2007), who argued that the 'food miles' concept (Paxton, 1994) was inaccurate and misleading in an environmental context.

2.3.5.2 Sensitivity analyses

Replacing the economic allocation method with the mass allocation method for all feed crops resulted in GWP and EP increasing by 6%, and AP increasing by 5%. These changes are due to the relatively low economic values (per a given mass) associated to the crop co-products used for pig feed. However, the new output values were largely proportional to the original values and did not affect the relative ranking amongst the three representative farms.

The inclusion of LUC for all crops increased GWP by 78%-81% from the baseline results. These changes were predominantly driven by land transformation from forest to arable land, including CO_2 emissions of 13,902 kg ha^{-1} for Argentinian soybean production. In the purely local context, because of the relative scale of the Argentinean soybean sector compared to the Irish pig sector, any increase in soybean demand in RoI will more likely be met by destination switch or perhaps altered crop choice rather than development of new arable land. On the global scale, however, the above finding supports the argument by Meul et al. (2012) that, in order to reduce carbon footprints, pig feed producers around the world should minimise LUC sensitive crop ingredients, such as soybean, by adopting low CP diets.

As discussed, the baseline result accounted for reduced production of N fertiliser owing to pig manure. When fertiliser offsetting was excluded from the model, GWP rose by 12-13 % from the corresponding baseline results. Exclusion of avoided NH_3 and NO_3^- increased AP and EP by 4% and 1%, respectively. While these results are sensitive to the soil type and weather and, therefore, cross-site comparisons are not straightforward, the values above are in line with other studies adopting similar approaches (Nguyen et al., 2011, Reckmann et al., 2013).

Finally, changing the assumption regarding on-farm energy usage using the upper and lower limit values reported by McCutcheon (2012) did not greatly affect the baseline results. High energy use resulted in a 2-3% increase in GWP, while the increases in AP and EP were even smaller (1% and < 1%, respectively). Low energy usage decreased GWP by 1-2%, with little effect (< 1%) observed for AP and EP. These findings suggest that the environmental footprint of pig production systems is not sensitive to the farm's strategy about energy usage.

2.3.6 Comparisons with previous research and system boundaries

A recent review by the author of the thesis categorised pig LCAs into three themes: feed, whole-system, or waste (McAuliffe et al., 2016). Of these three themes, **Table 2.10** offers a comparison of the current results with 14 other whole-system studies. Reviewing numerous LCA studies conducted in the area of food production, Roy et al. (2009) posited that cross-study comparisons are difficult due to different model assumptions and system boundaries. Indeed, some studies set the system boundary to the farm gate, while others include the abattoir **(Table 2.10)**. To navigate this limitation, the current study adopted two functional units (CW and LW), which allowed a broader interpretation of results. For example, Dourmad et al. (2014) report similar values of GWP, AP and EP for France using LW as the functional unit, whereas the GWP and AP values estimated by Nguyen et al. (2011) for Denmark based on CW are also comparable to the present study. Furthermore, while Nguyen et al. (2011) and Halberg et al. (2010) adopt a different metric for EP (g NO_3-eq), when the baseline EP from the current study is recalculated according to the same impact assessment method (Wenzel et al., 1997), the result (310 g NO_3/kg CW) is only slightly higher than the value reported by Nguyen et al. (2011) and slightly lower than that by Halberg et al. (2010) **(Table 2.10)**. It is therefore plausible to conclude that the environmental performance of the Irish pig sector is largely in line with wider European systems, including UK systems previously studied by Williams et al. (2006). Within the present dataset, the relative performances of the three representative farms were largely unaffected by this change in functional unit, as only small percentages of the overall environmental footprint originate from the slaughtering process **(Tables A2.1-A2.6)**. Finally, it is worthwhile noting that a recent worldwide analysis of pig supply chains (MacLeod et al., 2013) predicted that GWP values for Western European systems were in a region above 6 kg CO_2-eq/kg CW, higher than many of the studies presented in Table 7. However, this discrepancy is largely attributable to the fact that MacLeod et al. (2013) fully (and, consequently, perhaps excessively) account for LUC from soybean cultivation, rather than different functional units or system boundaries.

2.3.7 The global context

It is estimated that as much as 36% of energy produced by the world's crops are being used for animal feed, of which only 12% subsequently enter the human diet (Cassidy et al., 2013). Discussing necessary steps to realise global food security, Eisler et al. (2014) asserted the need to replace human-edible crops currently consumed by ruminants with human-inedible feeds such as grasses and pasture legumes. This challenge has an immediate and direct consequence on monogastric livestock systems around the world, which cannot necessarily adopt the same strategy to improve their production efficiency.

Previous research has shown that environmentally focused inclusion of SAA to feed formula can further reduce CP requirements in pigs through a targeted delivery of essential amino acids to counteract basal diet deficiencies (Osada et al., 2011). This reduction in CP is associated with lower GWP, AP and EP at both the feed production stage and during manure management, and likely creates further opportunities for improved environmental efficiencies (Garcia-Launay et al., 2014, Mosnier et al., 2011, Ogino et al., 2012). Regarding waste management, seemingly the most promising technology for reducing environmental impacts is anaerobic digestion of manures (McAuliffe et al., 2016). However, in order to make the system feasible at the global scale, issues such as the shortage of digestion plants and unappealing tariffs for selling energy back to the public grid must first be addressed (Nolan et al., 2012).

2.4 Conclusion

In this chapter, a common LCA method was applied to commercial pig production in RoI. For the average representative farm, GWP, AP and EP were estimated to be 3.51 kg CO_2-eq kg CW^{-1}, 43.8 g

SO_2-eq kg CW^{-1} and 32.1 g PO_4-eq kg CW^{-1}, respectively. Economically efficient herds demonstrated environmental improvements of up to 6% for GWP, 12% for AP and 15% for EP. Feed produced by a small-scale on-farm mill resulted in a lower GWP primarily due to more extensive usage of wheat and barley (rather than maize bran which required further processing), while AP and EP were elevated as a result of higher CP contents.

The results presented here suggest that improvements in on-farm production efficiency will generally also improve environmental sustainability of livestock production systems. However, further research is required to elucidate the exact nature of this correlation, and particularly how uncertainties governing physical, chemical and biological processes on the farm can potentially affect conclusions. As Chapter 3 shifts the focus from the intensive pork sector to the semi-extensive beef sector, a novel approach to LCA will be developed to consider these uncertainties in a more accurate manner.

Table 2.1. Feed composition for pig diets.

Ingredient (kg/1000 kg)	Origin[a]	Baseline analysis				Scenario analysis (on-farm feed mill)			
		Dry sow	Lact. sow	Weaner	Finisher	Dry sow	Lact. sow	Weaner	Finisher
Barley	IE	210	240	180	240				
	UK					350	320	350	362
Beet pulp	FR					80	20	20	25
Maize	FR	220	220	230	255	60	80	120	150
Premix	UK	25	25	25	20	28	40	35	28
Rapeseed meal	DE	70	30	40	85				
Soybean hulls	AR	50			15	50			
Soybean meal	AR	90	200	220	120	143	195	242	165
Soybean oil	AR	25	35	35	25	5	25	26	
Wheat	IE	44	80	108	58				
	FR	200	50		95				
	UK	66	120	162	87				
	DK					284	320	207	270

[a] IE: Ireland, UK: United Kingdom, FR: France, DE: Germany, DK: Denmark, AR: Argentina.

Table 2.2. Nutritional composition of individual feed ingredients (FAO, 2015), crop yields in primary production, and transportation distances.

Ingredient	Origin[a]	DM[b] (%)	CP (%)	P (g kg DM^{-1})	K (g kg DM^{-1})	Yield (kg DM ha^{-1})	Sea distance (km)	Road distance (km)[c]
Barley	IE	87.1	11.8	3.9	5.7	7050		93
	UK	87.1	11.8	3.9	5.7	5710	1413	88
Beet pulp	FR[d]	89.2	9.3	1	4.5	8920	832	229
Maize	FR	86.3	9.4	3	3.9	9030	832	145
Rapeseed meal	DE	91	34.1	11.5	12.5	3750	1428	319
Soybean hulls	AR	89.1	13.2	1.6	13.7	2440	11647	379
Soybean meal	AR	87.9	51.8	6.9	23.7	2440	11647	379
Wheat	IE	87	12.6	3.6	4.6	8570		22
	FR	87	12.6	3.6	4.6	6980	832	408
	UK	87	12.6	3.6	4.6	7480	454	374
	DK[4]	87	12.6	3.6	4.6	7160	2134	336

[a] IE: Ireland, UK: United Kingdom, FR: France, DE: Germany, DK: Denmark, AR: Argentina.

[b] DM: dry matter; CP: crude protein; P: phosphorus; K: potassium.

[c] Based on the distances between the largest arable region for the crop in each country (e.g. Cordoba for Argentinian soybean).

[d] These crop-origin combinations are used by the on-farm feed mill only.

Table 2.3. Performance data for three levels of productivity.

Parameter	Unit	AVG[a]	T25[b]	T10[c]
Breeding herd				
Sows	n	752	752	752
Replacement rate	%	50	52	48
Gilts	n	411	415	385
Sow mortality	%	5.1	3.7	3.7
Total litters per sow	n	4.3	4.4	4.7
Piglets per litter	n	13	13	13
Empty days	d	14	9.0	7.0
Sow liveweight	kg	250	250	250
Sow carcase yield	%	69	69	69
Feed consumed as dry sow	kg	1930	1980	2075
Feed consumed as lactating sow	kg	422	451	480
Feed consumed as gilt	kg	345	357	375
Growing pigs				
Weaning weight	kg	7.0	7.0	7.0
Weaner mortality	%	2.6	1.8	1.2
Feed consumed per weaner	kg	55	55	49
Finisher culling weight	kg	106	108	108
Finisher mortality	%	2.4	2.0	1.5
Finisher carcase yield	%	76	77	76
Feed consumed per finisher	kg	195	175	180
Total growing period	d	176	172	175

[a] Average herd efficiency
[b] Top 25% of Irish pig herds
[c] Top 10% of Irish pig herds.

Table 2.4. Life cycle inventory for slaughterhouse process.

	Unit	AVG[a]	T25[b]	T10[c]
Inputs				
Liveweight	kg	1000	1000	1000
Electricity	kWh	251.6	248.4	248.9
Diesel	kg	7.5	7.4	7.4
Transport	km	50	50	50
Outputs		0.0		
Carcase weight	kg	762.4	763.4	761.7
Losses				
CO	g	2.8	2.8	2.8
CO_2	kg	42.6	42.1	42.1
NO_x	g	28.2	27.8	27.9
N_2O	g	0.8	0.7	0.7
CH_4	g	0.8	0.8	0.8
BOD_5	g	888.9	877.8	879.5
COD	kg	23.1	22.8	22.9
N	kg	3.0	3.0	3.0
P	g	266.6	263.3	263.7
Biodegradable waste	kg	3.8	3.7	3.7

[a] Average herd efficiency
[b] Top 25% of Irish pig herds
[c] Top 10% of Irish pig herds

Table 2.5. Emission factors adopted in the current study.

Pollutant	Emission factor	Reference
CH_4		
Enteric fermentation (kg CH_4 head^{-1} year^{-1})		Duffy et al. (2017)
Gilts (in pig)	2.9	
Gilts (not served)	2.2	
Sows (in pig)	3.7	
Other sows	3.8	
Growing pigs > 20 kg	1.1	
Growing pigs < 20 kg	0.2	
Manure management (kg CH_4 head^{-1} year^{-1})		
Gilts (in pig)	8.0	
Gilts (not served)	5.0	
Sows (in pig)	8.0	
Other sows	18.8	
Growing pigs > 20 kg	5.1	
Growing pigs < 20 kg	3.4	
Direct N_2O-N		
Manure management		
In-house storage	0.002 x kg manure N ex-animal	IPCC (2006)
Outside storage with natural crust	0.005 x kg manure N ex-housing	
Field application	0.01 x kg manure N ex-storage	
Fertiliser application	0.01 x kg fertiliser N	
NO_x-N		
Manure management		
In-house storage	0.002 x kg manure N ex-animal	Dämmgen and Hutchings (2008)
Outside storage	0.005 x kg manure N ex-housing	
Field application	0.001 x kg manure N ex-storage	Nemecek and Kägi (2007)
Fertiliser application	0.007 x kg fertiliser N	(EEA, 2007)
NH_3-N		
Manure management		
In-house storage	0.13 x kg manure N ex-animal	Nguyen et al. (2010)
Outside storage	0.02 x kg manure N ex-housing	
Field application	0.07 x kg manure N ex-storage	Andersen et al. (2001)
After field application	0.117 x kg manure N ex-storage	Hansen et al. (2008)
Fertiliser application	0.065 x kg fertiliser N	Nguyen et al. (2010b)
NO_3-N leaching potential	kg N ex-animal - kg N total N loss - kg fertiliser N substitution	Nutrient balance
PO_4-P leaching potential	kg P ex-animal - kg fertiliser substitution	
Indirect N_2O-N	0.01 x kg (NH_3-N + NO_x-N) loss + 0.0075 x kg NO_3-N	IPCC (2006)

Table 2.6. LCI inputs and outputs for 1000 kg LW at the farm gate.

		Baseline analysis			Scenario analysis (on-farm feed mill)		
Item	Unit	AVG[a]	T25[b]	T10[c]	AVG[a]	T25[b]	T10[c]
Feed use	kg						
Dry sow		339	326	308	339	326	308
Lactating sow		74	70	71	74	70	71
Gilt		61	59	56	61	59	56
Weaner		517	514	453	517	514	453
Finisher		1790	1590	1640	1790	1590	1640
Total		2781	2559	2528	2781	2559	2528
Transport of feed (from mill)							
By truck	Tkm	313	288	285	0	0	0
Energy use							
Electricity	kWh	137	135	136	137	135	136
Heat (oil)	kWh	121	120	120	121	120	120
On-farm emissions							
Methane	kg						
Enteric fermentation		5.0	5.0	5.0	5.0	5.0	5.0
Manure management		63	61	62	63	61	62
Nitrous oxide	g	301	258	249	446	390	376
Ammonia	kg	5.4	4.6	4.5	8.0	7.0	6.8
Nitrogen oxides	g	631	539	520	933	817	785
Manure utilisation							
Transport	Tkm	72	62	60	107	94	90
Spreading	MJ	152	130	125	224	196	189
Nitrous oxide	g	669	572	552	989	866	832
Ammonia	kg	5.8	4.9	4.8	8.6	7.5	7.2
Nitrogen oxides	g	84	72	69	124	108	104
Nitrate	kg	45	39	37	70	59	56
Phosphate	g	222	164	156	385	314	304
Avoided fertiliser production	kg						
from manure nitrogen		39	33	32	57	50	48
from manure phosphorus		11	8.0	8.0	19	15	17
from manure potassium		26	24	23	35	32	32
Avoided fertiliser application							
Spreading	MJ	15	13	13	23	20	19
Nitrous oxide	g	161	137	132	237	208	200
Ammonia	kg	0.8	0.7	0.7	1.2	1.0	1.0
Nitrogen oxides	g	235	201	194	347	304	292

[a] Average herd efficiency; [b] Top 25% of Irish pig herds; [c] Top 10% of Irish pig herds

Table 2.7. Uncertainty parameters adopted in Monte Carlo analysis.

Emission	Uncertainty	Distribution	Reference
CH_4 (enteric fermentation)	± 17%	Triangular	Duffy et al. (2017)
CH_4 (manure management)	± 19%	Triangular	
N_2O (all emissions)	2 (SD^2)	Lognormal	IPCC (2006)
NH_3 (all emissions)	± 21%	Triangular	Amon et al. (2016)
NO_X (all emissions)	2 (SD^2)	Lognormal	
NO_3	1.58 (SD^2)	Lognormal	Pedigree matrix
PO_4	1.58 (SD^2)	Lognormal	Muller et al. (2016)

Table 2.8. LCIA results for the baseline analysis expressed per 1 kg carcase weight (CW) for three different levels of productivity.

	AVG[a]				T25[b]				T10[c]			
	Feed	Farm	Slaughter	Total	Feed	Farm	Slaughter	Total	Feed	Farm	Slaughter	Total
GWP (kg CO_2-eq kg CW^{-1})	2.03 (58%)	1.17 (33%)	0.31 (9%)	3.51	1.86 (56%)	1.14 (35%)	0.30 (9%)	3.30	1.85 (56%)	1.14 (35%)	0.31 (9%)	3.30
AP (g SO_2-eq kg CW^{-1})	19.5 (45%)	23.2 (53%)	1.1 (2%)	43.8	17.8 (46%)	20.0 (51%)	1.1 (3%)	38.9	17.7 (46%)	19.3 (51%)	1.1 (3%)	38.1
EP (g PO_4-eq kg CW^{-1})	16.2 (50%)	11.8 (37%)	4.1 (13%)	32.1	14.8 (51%)	10.2 (35%)	4.0 (14%)	29.0	14.7 (51%)	9.8 (34%)	4.1 (14%)	28.6

[a] Average herd efficiency

[b] Top 25% of Irish pig herds

[c] Top 10% of Irish pig herds

Table 2.9. Percentage differences in LCA outputs of scenario and sensitivity analyses relative to the baseline results.

	AVG[a]			T25[b]			T10[c]		
	GWP	AP	EP	GWP	AP	EP	GWP	AP	EP
On-farm feed mill	-11.1	13.5	6.5	-10.9	13.9	6.6	-10.9	13.1	5.9
Domestic wheat and barley	-1.1	0.2	-0.6	-0.9	0.0	-0.7	-0.9	0.3	-0.7
Mass allocation	6.3	5.3	6.5	6.1	5.1	6.6	6.1	5.5	6.6
Inclusion of land use change	80.9	-	-	79.7	-	-	77.9	-	-
High on-farm energy usage	3.1	0.9	0.6	3.3	1.0	0.7	3.3	1.0	0.7
Low on-farm energy usage	-2.0	-0.5	-0.3	-1.8	-0.5	-0.7	-2.1	-0.5	-0.7
Exclusion of fertiliser offsetting	13.1	4.3	1.6	11.8	4.1	1.4	11.5	4.2	1.4

[a] Average herd efficiency

[b] Top 25% of Irish pig herds

[c] Top 10% of Irish pig herds

Table 2.10. Comparisons of the present results with previous pig LCA studies.

Study	Scope	Functional unit	GWP	AP	EP
Basset-Mens and van der Werf (2005)	Crop production to pig farm gate	1 kg liveweight	2.3 kg CO_2-eq	43.5 g SO_2-eq	20.8 g PO_4-eq
Williams et al. (2006)	Crop production to pig farm gate	1000 kg carcase weight	6400 kg CO_2-eq	394 kg SO_2-eq	100 kg PO_4-eq
Dalgaard et al. (2007)	Crop production to delivery of pork to Port Harwich in Britain	1 kg pork	3.6 kg CO_2-eq	45 g SO_2-eq	232 g NO_3-eq
Perez (2009)	Crop production to pig farm gate	1000 kg liveweight	3284.3 kg CO_2-eq	43.8 kg SO_2-eq	192.6 NO_3-eq
Wiedemann et al. (2010)	Crop production to slaughterhouse	1 kg carcase weight at the meat processor gate	5.5 kg CO_2-eq	N/A	N/A
Halberg et al. (2010)	Crop production to pig farm gate	1 kg liveweight	3320 g CO_2-eq	61.4 g SO_2-eq	381 g NO_3-eq
Nguyen et al. (2010)	Crop production to pig farm gate	1 kg slaughter weight	4812 g CO_2-eq	N/A	N/A
Pelletier et al. (2010a)	Crop production to pig farm gate	1 kg liveweight	2.5 kg CO_2-eq	N/A	15.9 g PO_4-eq
Nguyen et al. (2011)	Crop production to slaughterhouse gate	1 kg pork delivered from the slaughterhouse	3.1 kg CO_2-eq	56 g SO_2-eq	243 g NO_3-eq
Devers et al. (2012)	Crop production to delivery of pork to Antwerp in Belgium	1 kg carcase weight	2.6 kg CO_2-eq	39 g SO_2-eq	22 g PO_4-eq
Dolman et al. (2012)	Crop production to pig farm gate	100 kg liveweight	546 kg CO_2-eq	5.3 kg SO_2-eq	61.4 kg NO_3-eq
Jacobsen et al. (2014)	Crop production to meat processor gate	1 kg deboned pigmeat	4.8 kg CO_2-eq	N/A	N/A
Reckmann et al. (2013)	Crop production to slaughterhouse gate	1 kg pork slaughter weight	3.2 kg CO_2-eq	57.1 g SO_2-eq	23.3 PO_4-eq
Dourmad et al. (2014)	Crop production to pig farm gate	1 kg liveweight	2.3 kg CO_2-eq	44 g SO_2-eq	18.5 PO_4-eq
Current study	Crop production to pig farm gate	1 kg liveweight	2.4 kg CO2-eq	32.6 g SO2-eq	21.4 g PO4-eq
Current study	Crop production to slaughterhouse gate	1 kg carcase weight	3.5 kg CO2-eq	43.8 g SO2-eq	32.1 g PO4-eq

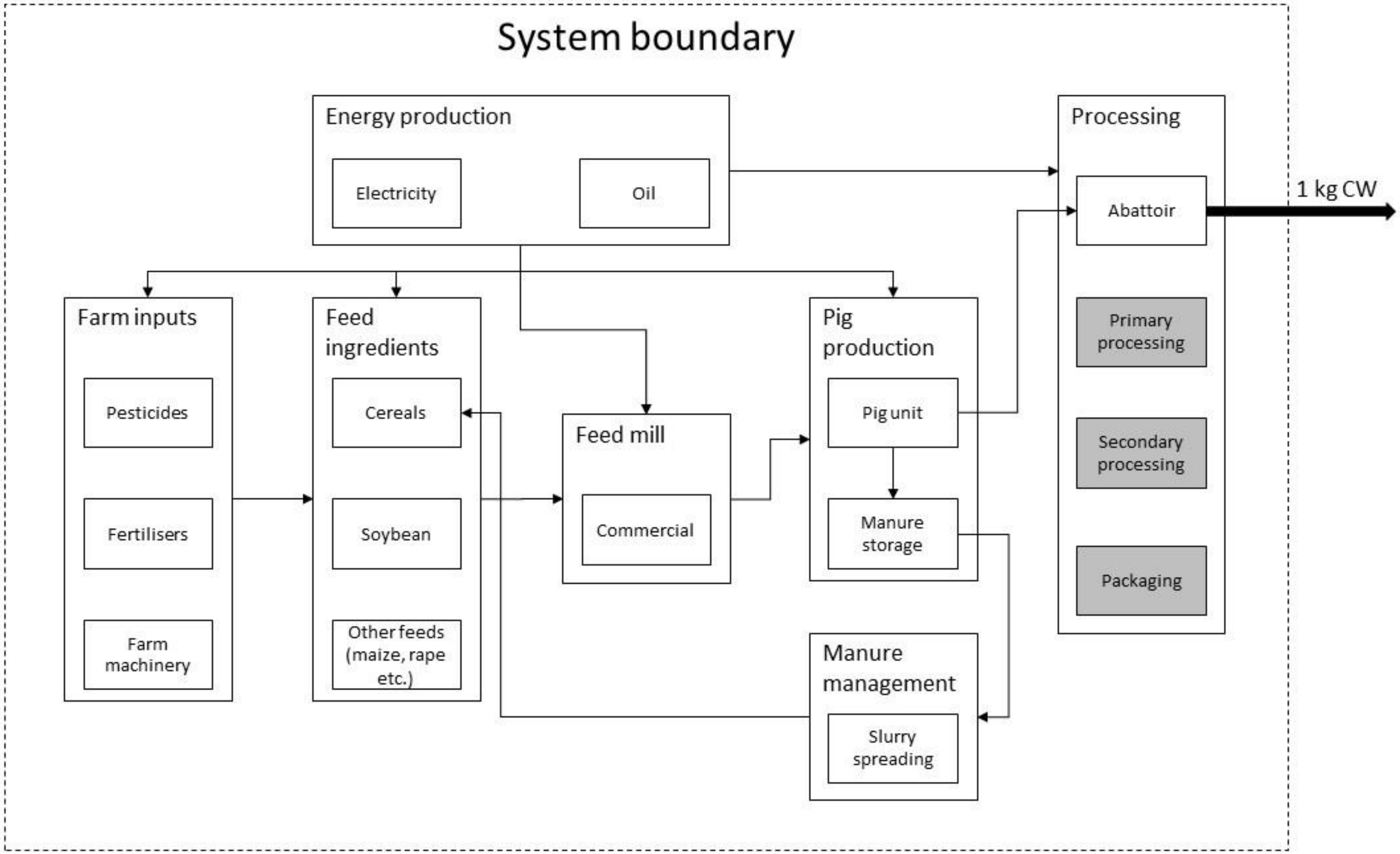


Figure 2.1. Stylised schematic of the baseline study boundary. Grey processes are excluded from analysis.

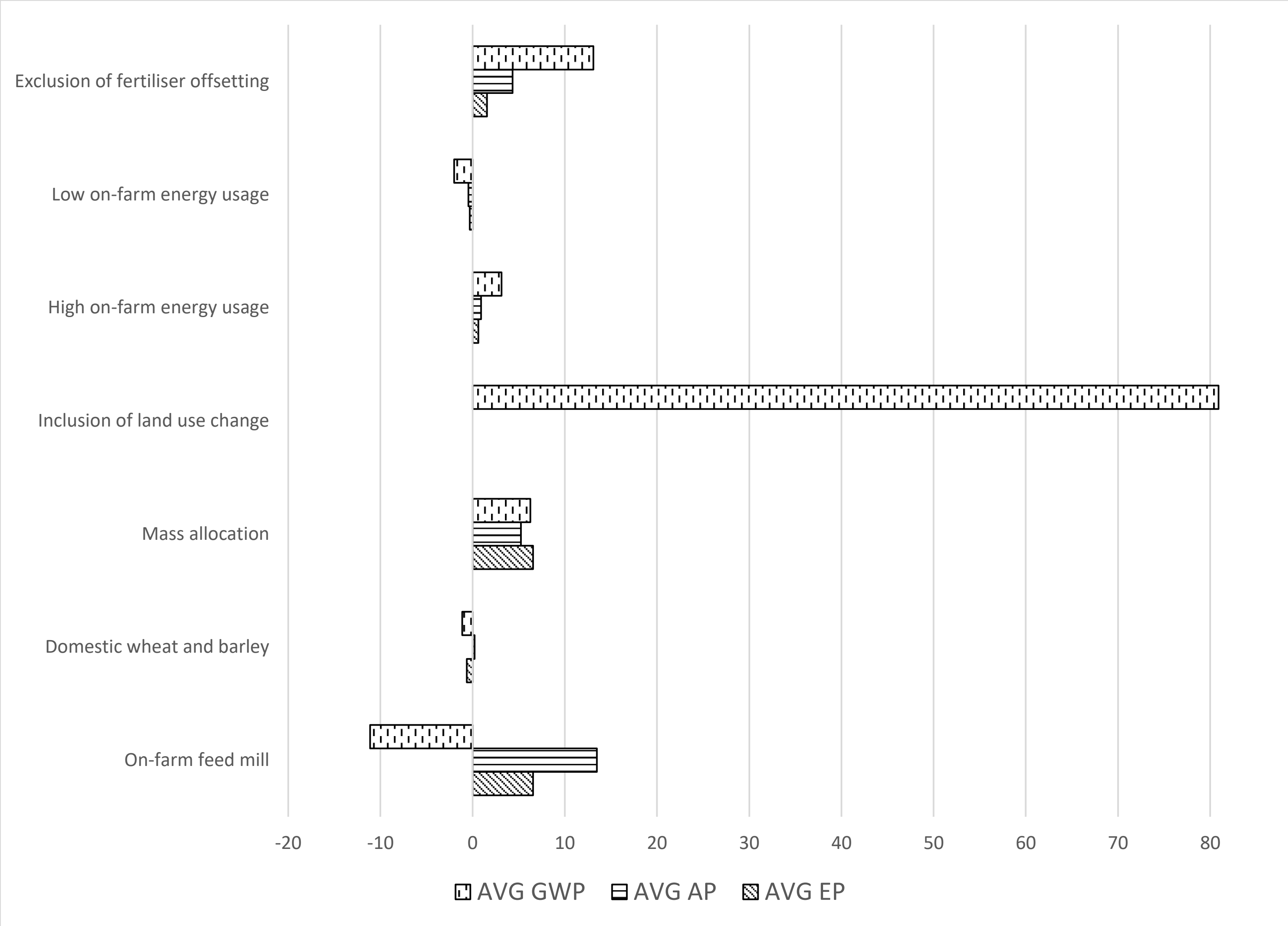


Figure 2.2. Effect of different analyses on baseline results, presented as percentage change. AVG: average herd performance; GWP: global warming potential; AP: acidification potential; EP: eutrophication potential.

Appendix to Chapter 2.

Table A2.1. Environmental impacts for the average (AVG) farm based on the baseline analysis.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	237	2.3	2
Lactating sow	54.6	0.5	0.4
Gilt	42.4	0.4	0.4
Weaner	388	3.4	2.9
Finisher	1310	12.7	10.5
Transport of feed from mill to farm			
By truck	29.5	0.1	< 0.1
Energy use			
Electricity	130	0.5	0.4
Heat (oil)	51	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	178.5	-	-
Manure management	813.1	-	-
Nitrous oxide	118	-	< 0.1
Ammonia	-	11.3	2.5
Nitrogen oxides	-	0.4	< 0.1
Manure utilisation			
Transport	36	0.3	< 0.1
Spreading	15.9	< 0.1	< 0.1
Nitrous oxide	262	-	
Ammonia	-	12.2	2.7
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	5.9
Phosphate	-	-	0.3
Avoided fertiliser production			
from manure nitrogen	-334	-	-
from manure phosphorus	-39.4	-	-
from manure potassium	-27.4	-	-
Avoided fertiliser application			
Spreading	-1.6	- < 0.1	- < 0.1
Nitrous oxide	-62.8	-	- < 0.1
Ammonia	-	-1.7	-0.4
Nitrogen oxides	-	-0.2	- < 0.1
Slaughterhouse			
Electricity	239	1	0.6
Diesel	5.7	< 0.1	< 0.1
Transport	3.9	< 0.1	< 0.1
Landfill	2.5	< 0.1	< 0.1
Emissions (aggregated)	50	0.1	3.4

Table A2.2. Environmental impacts for the T25 farm based on the baseline analysis.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	227	2.2	1.9
Lactating sow	51.1	0.5	0.4
Gilt	41	0.4	0.3
Weaner	385	3.5	2.9
Finisher	1160	11.2	9.3
Transport of feed from mill to farm			
By truck	27	0.1	< 0.1
Energy use			
Electricity	128	0.5	0.3
Heat (oil)	50.3	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	171.6	-	-
Manure management	780.6	-	-
Nitrous oxide	100.7	-	< 0.1
Ammonia	-	9.7	2.2
Nitrogen oxides	-	0.3	< 0.1
Manure utilisation			
Transport	30.7	0.2	< 0.1
Spreading	13.6	< 0.1	< 0.1
Nitrous oxide	223.8	-	0.1
Ammonia	-	10.4	2.2
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	5.1
Phosphate	-	-	0.2
Avoided fertiliser production			
from manure nitrogen	-285	-	-
from manure phosphorus	-29	-	-
from manure potassium	-24.9	-	-
Avoided fertiliser application			
Spreading	-1.4	- < 0.1	- < 0.1
Nitrous oxide	-53.6	-	- < 0.1
Ammonia	-	-1.5	-0.3
Nitrogen oxides	-	-	- < 0.1
Slaughterhouse			
Electricity	236	1	0.6
Diesel	5.6	< 0.1	< 0.1
Transport	3.8	<0.1	< 0.1
Landfill	2.5	< 0.1	< 0.1
Emissions (aggregated)	60	0.1	3.4

Table A2.3. Environmental impacts for the T10 farm based on the baseline analysis.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	216	2.1	1.8
Lactating sow	52.5	0.5	0.4
Gilt	39	0.4	0.3
Weaner	340	3.1	2.6
Finisher	1200	11.6	9.6
Transport of feed from mill to farm			
By truck	26.8	0.1	< 0.1
Energy use			
Electricity	129	0.5	0.3
Heat (oil)	50.6	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	171.3	-	-
Manure management	789.6	-	-
Nitrous oxide	97.4	-	< 0.1
Ammonia	-	9.4	2
Nitrogen oxides	-	0.3	< 0.1
Manure utilisation			
Transport	29.7	0.2	< 0.1
Spreading	13.2	< 0.1	< 0.1
Nitrous oxide	215.8	-	0.1
Ammonia	-	10.1	2.2
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	4.9
Phosphate	-	-	0.2
Avoided fertiliser production			
from manure nitrogen	-276	-	-
from manure phosphorus	-27.7	-	-
from manure potassium	-24.5	-	-
Avoided fertiliser application			
Spreading	-1.3	- < 0.1	- < 0.1
Nitrous oxide	-51.8	-	- < 0.1
Ammonia	-	-1.4	-0.3
Nitrogen oxides	-	-0.1	- < 0.1
Slaughterhouse			
Electricity	237	1	0.6
Diesel	5.6	< 0.1	< 0.1
Transport	3.9	< 0.1	< 0.1
Landfill	2.5	< 0.1	< 0.1
Emissions (aggregated)	60	< 0.1	3.4

Table A2.4. Environmental impacts for the AVG farm when feeds are sourced from the on-farm mill.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	180	1.6	1.4
Lactating sow	44.8	0.4	0.3
Gilt	32.2	0.3	0.3
Weaner	341	2.8	2.4
Finisher	1090	9.7	8.3
Transport of feed from mill to farm			
By truck	-	-	-
Energy use			
Electricity	130	0.5	0.4
Heat (oil)	51	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	178.5	-	-
Manure management	813.1	-	-
Nitrous oxide	174.1	-	0.1
Ammonia	-	16.9	3.7
Nitrogen oxides	-	0.6	0.1
Manure utilisation			
Transport	53.1	0.4	< 0.1
Spreading	23.5	0.1	< 0.1
Nitrous oxide	386	-	0.3
Ammonia	-	17.9	4
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	8.8
Phosphate	-	-	0.5
Avoided fertiliser production			
from manure nitrogen	-493	-	-
from manure phosphorus	-68.2	-	-
from manure potassium	-37.4	-	-
Avoided fertiliser application			
Farm traction	-2.4	- < 0.1	- < 0.1
Nitrous oxide	-92.8	-	- < 0.1
Ammonia	-	-2.5	-0.5
Nitrogen oxides	-	-0.2	- < 0.1
Slaughterhouse			
Electricity	239	1	0.6
Diesel	5.7	< 0.1	- < 0.1
Transport	3.9	< 0.1	- < 0.1
Landfill	2.5	< 0.1	- < 0.1
Emissions (aggregated)	60	< 0.1	3.4

Table A2.5. Environmental impacts for the T25 farm when feeds are sourced from the on-farm mill.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	173	1.6	1.4
Lactating sow	41.9	0.4	0.3
Gilt	31.1	0.3	0.2
Weaner	339	2.8	2.4
Finisher	967	8.6	7.4
Transport of feed from mill to farm			
By truck	-	-	-
Energy use			
Electricity	128	0.5	0.3
Heat (oil)	50.3	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	171.6	-	-
Manure management	780.6	-	-
Nitrous oxide	152.2	-	< 0.1
Ammonia	-	14.7	3.2
Nitrogen oxides	-	0.6	< 0.1
Manure utilisation			
Transport	46.5	0.3	< 0.1
Spreading	20.6	0.1	< 0.1
Nitrous oxide	338	-	0.3
Ammonia	-	15.7	3.5
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	7.7
Phosphate	-	-	0.4
Avoided fertiliser production			
from manure nitrogen	-431	-	-
from manure phosphorus	-55.6	-	-
from manure potassium	-34.1	-	-
Avoided fertiliser application			
Farm traction	-2.1	- < 0.1	- < 0.1
Nitrous oxide	-81.1	-	- < 0.1
Ammonia	-	-2.2	-0.5
Nitrogen oxides	-	-0.2	- < 0.1
Slaughterhouse			
Electricity	236	1	0.6
Diesel	5.6	< 0.1	< 0.1
Transport	3.8	< 0.1	< 0.1
Landfill	2.5	< 0.1	< 0.1
Emissions (aggregated)	50	< 0.1	3.3

Table A2.6. Environmental impacts for the T10 farm when feeds are sourced from the on-farm mill.

Item	GWP (g CO_2-eq)	AP (g SO_2-eq)	EP (g PO_4-eq)
Feed production and transportation			
Dry sow	164	1.5	1.3
Lactating sow	43.1	0.4	0.3
Gilt	29.6	0.3	0.2
Weaner	299	2.4	2.1
Finisher	1000	8.9	7.7
Transport of feed from mill to farm			
By truck	-	-	-
Energy use			
Electricity	129	0.5	0.3
Heat (oil)	50.6	0.1	< 0.1
Farm emissions			
Methane			
Enteric fermentation	171.3	-	-
Manure management	789.6	-	-
Nitrous oxide	146.4	-	< 0.1
Ammonia	-	14.2	3.1
Nitrogen oxides	-	0.5	< 0.1
Manure utilisation			
Transport	44.8	0.3	< 0.1
Spreading	19.8	< 0.1	< 0.1
Nitrous oxide	326	-	0.3
Ammonia	-	15.2	3.3
Nitrogen oxides	-	< 0.1	< 0.1
Nitrate	-	-	7.4
Phosphate	-	-	0.4
Avoided fertiliser production			
from manure nitrogen	-416	-	-
from manure phosphorus	-53.9	-	-
from manure potassium	-33.6	-	-
Avoided fertiliser application			
Farm traction	-2	- < 0.1	- < 0.1
Nitrous oxide	-78.1	-	- < 0.1
Ammonia	-	-2.1	-0.5
Nitrogen oxides	-	-0.2	- < 0.1
Slaughterhouse			
Electricity	237	1	0.6
Diesel	5.6	< 0.1	< 0.1
Transport	3.9	< 0.1	< 0.1
Landfill	2.5	< 0.1	< 0.1
Emissions (aggregated)	50	< 0.1	3.4

Chapter 3 - Distributions of emissions intensity for individual beef cattle reared on pasture-based production systems

3.1 Introduction

When the LCA method is applied at the farm scale, life cycle inventories of representative farms are often constructed from a steady-state herd structure using national statistics or, less frequently, farm surveys (Wiedemann et al., 2015); the analysis carried out in Chapter 2 was an example of the former approach. As information on individual animals is often unavailable from these datasets, livestock data may already be aggregated at the time of inventory analysis, both across individual animals and across seasons. Although uncertainty analyses, such as those outlined in Section 2.2.3, can be seen as a tool to partially account for genetic and seasonal variabilities in livestock-originated carbon footprints, the degree to which these methods can address the bias suffered by these representative animal approaches is not well-understood.

Using primary data collected on the North Wyke Farm Platform (NWFP) in Devon, UK, of which details were described in Section 1.4, this chapter proposes a novel approach of life cycle impact assessment that can explicitly account for heterogeneity in animal performance, both individually and seasonally. Field data were measured at high spatial and temporal resolutions, enabling a unique research platform to conduct a detailed analysis of environmental hotspots. The use of individual animal data allowed computation of emissions intensity for each growing calf and, by extension, their intra-farm distributions, offering an alternative method to the Monte Carlo analysis whereby livestock performance parameters are assumed to follow distributions pre-identified based on best-available external data. More specifically, the objective of this particular study was to explore the potential benefit of this new approach with regards to livestock-originated uncertainty; methods to reduce pasture-originated uncertainty will be discussed in Chapters 4 and 5. Here, emissions intensity for pasture-based beef finishing systems at the NWFP was quantified under two methods, namely with and without information on individual animal performance. To the best of author's knowledge, this is the first environmental assessment of the English cattle industry based on high-resolution primary data; the industry employs 440,000 people and is estimated to be worth £2.8 billion, with high dependence on grazed systems (Marsh et al., 2012).

3.2 Materials and methods

Similarly to Chapter 2, this study follows the standard LCA protocol to evaluate emissions intensity, focusing here on pasture-based cattle production systems. However, as the study only entails the post-weaning (finishing) stage of the cattle lifecycle (for reasons discussed below), it is not a full carbon footprint analysis in the strictest sense as defined by PAS 2050 guidelines (BSI, 2011).

3.2.1 System boundary and functional unit

A schematic diagram of the system boundary is provided in **Figure 3.1**. The present study adopts a "gate to gate" approach (Berton et al., 2016, Ogino et al., 2004), whereby all non-capital inputs and outputs related to the post-weaning phase of cattle production are included in the model. Production processes for farm building infrastructure were excluded from the model following the approach adopted by recent studies (Buratti et al., 2017, Mogensen et al., 2015). Temporally, the present study follows emissions intensity of 90 cattle (30 per farmlet system) that were born in the spring of 2014. Thus, the on-farm component of **Figure 3.1** corresponds to the period from October 2014, when they were weaned from their mothers, to their departure to the slaughterhouse, around December 2015 for the majority of the animals, on meeting weight and carcase specification targets (Section 1.2).

As will be seen in Chapter 5, cow-calf operations generate a high proportion of the total carbon footprint associated with beef production, sometimes as much as two-thirds on pasture-based systems (Pelletier et al., 2010b). Notwithstanding, the suckler system was excluded from the present analysis for the following three reasons, primarily so that methodological comparison between old and new approaches (to be outlined below) could be carried out with the smallest possible set of confounding factors. First, by randomising the allocation of calves to each farmlet at weaning, factors related to *mothers'* body conditions that affect *calves'* performance early in life (e.g. quantity and quality of milk provided) are also randomised, and thus the difference in system-wide economic and environmental performance amongst the three finishing systems becomes fully attributable to their pasture management strategies. Second, as all animals are maintained on the same (external) permanent pasture system prior to their entrance to the NWFP, excluding this stage from the computation of lifecycle emissions intensity does not affect the relative ranking of the three systems. Third, and most importantly, as the cow-calf operation in North Wyke is not part of the NWFP, it does not record field data at a resolution comparable to what is collected on the NWFP. While it is possible to estimate emissions intensity of the suckler operation based on a combination of low-resolution data and published equations (as will be done in Chapter 5), doing so would likely compromise the accuracy of the methodological comparison and therefore was judged to be undesirable in this instance. This decision was made at the cost of reduced comparability of results with previous beef LCA research, many of which operate under a wider system boundary. For this reason, inter-study comparisons of carbon footprints will be conducted in Chapter 5.

Given that the entire lifecycle of cattle was not examined, the more common functional units for beef LCA studies, such as liveweight (Ridoutt et al., 2011) or carcase weight (Mogensen et al., 2015, Peters et al., 2010) were inappropriate. Instead, the functional unit was set as "1 kg of liveweight gain (LWG)", a key indicator of the animal performance post-weaning that has previously been adopted by Casey and Holden (2006), Dick et al. (2015) and Ruviaro et al. (2015), amongst others. The use of this functional unit implies that the partitioning of an additional mass acquired by livestock (between muscle, fat and other parts of the body) is assumed not to differ considerably amongst individual animals.

3.2.2 Inventory analysis and impact assessment

As discussed, the majority of on-farm information utilised in the present study was collected in the form of primary data. Individual animals were weighed every two to four weeks using a cattle crush and weigh head, providing a high temporal resolution for average daily gains (ADG). During the grazing season, sward snip samples were collected in the same weeks when animals were weighed from all fields occupied by cattle at that time. These samples were cut at grazing height (5 cm above ground level) along a W-transect, ignoring dead material, seed heads and weeds which animals tend to avoid. During winter, grab samples of silage were collected at a similar frequency, from five points along the width of each barn feed passage during feeding time, so that they represented roughage consumed by livestock at that time. Samples were stored at -20°C until chemical analysis was carried out.

Modified Acid Detergent Fibre (MADF) composition for both pasture and silage samples was quantified using a FOSS Fibertec 8000 Auto Fiber Analysis System following the method of Clancy and Wilson (1966). Samples were freeze dried and then ground using a Cyclone Sample Mill so that material could pass through a 1 mm sieve. Following this preparation, 1000 ± 2 mg of sample was added to oven dried crucibles. Crucibles were first inserted to the Fibertec cold extraction unit to remove excessive fat content using 25 ml of acetone and then placed into the Fibertec hot extraction unit. Acid detergent solution (ADS) was made by mixing 0.5 M (1N) of H_2SO_4 with 20 g/l of CTAB (HexadecylTrimethylAmmonium Bromide 98%). Using this solution, modified acid detergent solution (MADS) was subsequently produced by mixing equal volumes of ADS and H_2SO_4. The hot extraction unit automatically distributed MADS and antifoaming agent (n-Octanol) to the samples. Following hot extraction, 25 ml of acetone was added to samples and drained. Once analysis was complete, the derived MADF fractions were converted to corresponding metabolisable energy (ME) values using equations independently calibrated for UK pastures and silages (Alderman and Cottrill, 1993). These values were further converted to digestible organic matter content (DOMD; reported hereafter as digestible energy) using a separate equation (Alderman and Cottrill, 1993).

Total N contents of feed were measured using an elemental analyser and isotope ratio mass spectrometer. Samples were weighed to 2 ± 0.1 mg using a Mettler Toledo MX5 electronic microbalance and inserted to 5 x 3.5 mm tin capsules. They were then analysed in a Carlo Erba NA2000 elemental analyser connected to a Sercon 20-22 isotope ratio mass spectrometer. The derived total N values were converted to crude protein content by multiplying the standard coefficient of 6.25 (FAO, 2003).

Detailed records of all farm inputs were maintained throughout the season. These include, for example, the type and amount of fertilisers and pesticides used, the areas these products were applied to, and supplementary feeds used during housing. **Table 3.1** provides a detailed breakdown of inputs applied to the NWFP during the temporal boundary of the study. Data for background processes such as production of fertiliser, supplementary feeds, bedding and seeds, were sourced from the *ecoinvent* database V3 (Wernet et al., 2016). Sea-based transportation distances were calculated using data from Portworld (Portworld, 2016), while road distances were calculated using a geographical information system (GIS) platform.

Emissions arising from livestock and pastures were calculated using a modified IPCC Tier 2 approach (IPCC, 2006). In order to examine both temporal differences of emissions and the effects of animal heterogeneity, livestock emissions were calculated for each animal for each time period, or between two weighing events, using the weighing records, digestible energy and crude protein values obtained in the methods described above. Calculations were programmatically automated and linked to the NWFP database so as to apply different parameters depending on the animal's age, location and feed being consumed **(Figures 3.2 and 3.3)**. This model design was motivated by an earlier finding that the difference in direct emissions between times when animals are on pasture and in housing is primarily driven by digestibility (affecting rumen methane production) and, to a lesser degree, crude protein content (affecting nitrogen-based emissions) of feed (Boadi and Wittenberg, 2002). For computation of CH_4 emissions through manure management, cattle manure collected during the winter housing period (Section 1.4) was assumed to be kept in middens for an average of six months.

As discussed in Chapter 1, sheep also occupy grassland of the NWFP as part of rotational grazing systems, although they do not share the same pasture with cattle at any given time. Considering that their manure also contributes to pasture growth (and thus indirectly facilitates cattle LWG) and *vice versa* (cattle manure facilitates sheep LWG), the entire environmental burdens originating from pastures were first split between the two enterprises based on economic values of products leaving the system boundary (i.e. economic allocation). The emissions allocated to the cattle operation were further split among individual animals under the rules that: (a) emissions originating from material inputs to pastures (e.g. inorganic fertilisers and use of machinery) and sheep manure were evenly distributed across 30 cattle on each system; and (b) emissions arising from cattle manure (minus those allocated to the sheep enterprise) were calculated individually for each cattle, taking animals' growth performance into consideration. All results reported below are net of GHG emissions attributable to sheep production.

Grasslands in the southwest of England are typically located on hilly land with soils that have the propensity to become supersaturated. As these lands are unsuitable for arable crop production, emissions owing to land use and land use change were not included in the present model. Similarly, given the small quantities of soybean (*Glycine max*) supplemented to the animals during the final weeks in housing **(Table 3.1)**, land use change (LUC) associated with the production of soybean was not considered. Finally, grasslands are sometimes credited as being net sinks of CO_2, although a large degree of uncertainties exists for these estimates (Beauchemin et al., 2011). Following PAS 2050 guidelines (BSI, 2011), the potential effect of changes in soil carbon stock on emissions intensity was not considered in this study, as reseeding of white clover (WC) and high-sugar grass (HS) treatments did not involve LUC. For the purpose of calculating environmental burdens associated with on-farm activities for reseeding **(Table 1.1)**, WC and HS systems were assumed to be renewed every five years. As will become clear, the results were not sensitive to this sowing interval.

On completion of the life cycle inventory, emissions intensity for each individual animal was estimated according to the IPCC 2013 100-year average method (IPCC, 2013) using SimaPro V8.2.3 (www.simapro.com). Under this method, CH_4 and N_2O are respectively assumed to have 30.5 and 265 times greater impacts than CO_2 on climate change. Processes were designed so the sum of emissions from all individual animals theoretically equates to the total emissions from the cattle-finishing enterprise of each farmlet.

3.2.3 Interpretation

Statistical interpretation was carried out using GenStat V17.1 (www.vsni.co.uk/software/genstat). Based on performance data and emissions intensity estimates for individual animals, multi-sample F-tests (one-way analysis of variance) and two-sample t-tests were conducted to examine differences in livestock performance and emissions intensity between the farmlets. Correlations between emissions intensity and its potential determinants were assessed using Pearson's correlation coefficient.

As discussed, estimation of emissions intensity for individual animals in this study was motivated by uncertainty inherent within life cycle data, which is regarded as one of the most limiting factors of the LCA framework (Groen and Heijungs, 2017). However, information on the performance of individual animals is not always available to LCA practitioners, especially outside the research farm environment. In order to examine the potential discrepancy in model outputs between these two situations, an alternative method of estimation was also set up as part of the analysis. Here, variables related to animal performance (e.g. ADG and days on farm) were averaged across the entire herd based on low temporal resolution (yearly) data, generating a single value of emissions intensity for a representative animal reared on each farmlet. Following this procedure, the effect of uncertainty was evaluated by means of a Monte Carlo analysis, and the resultant outputs were compared pairwise between the three farmlets. Furthermore, to evaluate the degree of interactions between the two methods of uncertainty analysis, a similar assessment was also carried out for the best and worst performing animals on each farmlet (as judged by emissions intensity) that were identified under the animal-by-animal approach. All Monte Carlo simulations were conducted using SimaPro V8.2.3, where parameters were randomly drawn over 1000 iterations from the distributions summarised in **Table 3.2**.

Finally, in line with ISO 14040 (ISO, 2006), a sensitivity analysis was conducted to test the effect of choosing the economic allocation method (for emissions from pastures) on the model outputs. The mass allocation approach was selected as an alternative, whereby the allocation ratio was determined by estimated dry matter intake (DMI) of cattle and sheep. In addition, a sensitivity test was also carried out to test the impact of having applied the IPCC 2013 conversion factors to derive CO_2-equivalent values for other GHG (30.5 for CH_4 and 265 for N_2O) *vis-à-vis* the superseded IPCC (2006) factors, which had considerably different specifications (25 for CH_4 and 298 for N_2O).

3.3 Results and discussion

3.3.1 Inter-system differences

Across the three systems, a short-term decrease in ADG was observed immediately post-weaning. As animals grew larger, their ADG increased to around 1.4-1.6 kg/d, until they reached a mature age and then slowed down to "finish" or meet conformation and fat scores **(Figure 3.4)**. The relatively low overall ADG compared to the common target rate in the study region (0.8-1.0 kg/d) was due to extended housing and difficulty in satisfying carcase specification criteria. A statistically significant difference in ADG was observed amongst the three systems, with the animals on PP growing faster than WC and HS **(Table 3.3)**. This result is largely attributable to their relative performance during the aforementioned conditioning period, while the degree of inter-system difference was considerably lower earlier in the season. As a result of randomised allocation and weight targeting, there were no significant inter-group differences for entry weight or finishing weight.

Table 3.4 displays the major contributors to total emissions intensity in each farmlet. In consonance with other LCA studies on beef production systems (as reviewed by de Vries et al., 2015), methanogenic emissions from the rumen were the single greatest source of GHG emissions irrespective of pasture management strategies. The WC system had the lowest average emissions intensity across all animals (16.0 kg CO_2-eq/kg LWG), a result primarily driven by lower requirements of inorganic N fertiliser, followed by PP (18.5 kg CO_2-eq/kg LWG) and HS (20.2 kg CO_2-eq/kg LWG). Multi-sample F-tests based on emissions intensity of individual animals showed there were significant differences across the three treatments ($p < 0.001$). Pairwise (via t-tests), emissions intensity for WC was significantly lower than PP ($p < 0.001$) and HS ($p < 0.001$), while PP was significantly lower than HS ($p < 0.001$). With regard to direct livestock emissions, the PP farmlet performed most favourably due to higher ADG **(Table 3.3)**. However, care should be taken at the interpretation of ADG; relatively low nutrient values for WC (crude protein) and HS (digestible energy) could be a reflection of the fact that WC and HS swards were close to establishment **(Table 3.1)**. Higher animal performance on PP notwithstanding, reduced N fertiliser usage on WC was by far the greatest saving to total GHG emissions across all systems although, over the long term, the negative impact of legumes on the soil carbon stock (Herridge and Brock, 2016) may also need to be considered.

Figure 3.5 displays the relationship between ADG and emissions intensity under each treatment. Strong and statistically significant negative correlations were found between the two variables for all three systems (r = -0.86, -0.84 and -0.77 respectively for PP, WC and HS; all $p < 0.001$), suggesting that the inter-system differences in mean emissions intensity values are, to a large degree, explained by differences in ADG. As for the reasons for differences in ADG, the slower growth rates by WC cattle can largely be attributed to the lower crude protein content in silage **(Table 3.1)**. For HS animals, the greater heterogeneity of monoculture swards at the height of the pasture growing season, which was consistently observed and, in this instance, resulted in a lower yield recorded from summer silage cuts **(Table 3.1)**, would likely have been a leading contributing factor, although the digestible energy of HS grass was also lower than expected (but statistically not different to PP). This hypothesis, in turn, seems consistent with smaller variances for both ADG and emissions intensity amongst PP animals, as seen in both **Table 3.4** and **Figure 3.5**. The above evidence indicates that, while the PP system has a higher emissions intensity than the WC system on average, it may possess a comparative advantage from the viewpoint of system stability and thus a less stringent requirement for animal selection, at least during early years of WC sward establishment following pasture renewals. To maximise the genetic potential of latest germplasm used by the WC and HS farmlets, strategies to reduce spatial variability of swards, such as spatial separation (Sharp et al., 2014), overseeding (Rouquette, 2016) and precision agriculture (Hedley, 2015), may need to be explored. Pasture designs that optimise the balance of nitrogen and energy release in the rumen, both at grazing and from silage, will increase ruminal microbial protein synthesis and subsequently animal performance (Lee et al., 2001, Merry et al., 2006).

3.3.2 Intra-system distributions

As the emissions associated with pasture were evenly distributed across 30 cattle on each farmlet, the observed intra-system variation in emissions intensity is solely attributable to individual livestock performance. A closer investigation of these distributions suggests that, although emissions from livestock were not the primary drivers of relative emissions intensity amongst different farmlets, individual animal heterogeneity played a key role in distributions of emissions intensity within each particular farming system **(Figure 3.6).** In addition to 33% (PP), 52% (WC) and 54% (HS) differences in emissions intensity between the best and worst performing animals on the farms **(Table 3.4)**, there were notable differences in animal performance between sexes under two of the three treatments **(Figure 3.7)**. Steers from the WC farmlet were found to have a significantly lower emissions intensity than WC heifers (difference in means = 1.4 kg CO_2-eq/kg LWG; p = 0.020 based on the paired t-test). Similarly, HS steers had a significantly lower emissions intensity than HS heifers (1.7 kg CO_2-eq/kg LWG; p = 0.027). While there were no significant differences in ADG between the sexes within either of these treatments, steers had higher total LWG than heifers for both WC and HS systems (p = 0.033 and p = 0.037, respectively). Interestingly, HS heifers spent significantly less time on the NWFP than steers (difference in means = 34 days; p = 0.010) because of their lower target weight and propensity for heifers to meet carcase specification requirements more easily. However, the associated savings in livestock-based emissions were not large enough to offset the benefits of larger total growth by steers. This finding reiterates the importance of considering interlinkages with external supply chains (Brock et al., 2013) and may support an argument for dairy beef production (in which more males are reared for meat than females) to create more sustainable livestock systems (de Vries et al., 2015), although the slower growth rate by dairy breeds, as well as the greater finishing potential of bulls, must also be taken into consideration in this debate. Further research is required before drawing any conclusion regarding the optimal interlinkages between beef systems and dairy systems, which is beyond the remit of the present study.

3.3.3 Methodological comparisons

As described earlier, emissions intensity was also computed for a pre-averaged representative animal on each farmlet. The resultant point estimates for emissions intensity under PP, WC and HS systems were 17.6, 14.3 and 18.8 kg CO_2-eq/kg LWG, respectively. Compared to the arithmetic means of emissions intensity values across individual animals **(Table 3.4)**, the alternative approach was found to underestimate the emissions intensity by 0.9–1.7 kg CO_2-eq/kg LWG, or up to 10% of system-wide emissions.

According to Monte Carlo pairwise comparisons carried out for these pre-averaged animals, the PP and HS systems both had significantly higher emissions intensity than the WC system (p = 0.017 and p = 0.001, respectively); however, there were no significant differences between the PP and HS systems (p = 0.293). This finding contrasts with the aforementioned t-test results based on emissions intensity derived for individual animals, whereby the mean values from all three systems were found to be significantly different. The reason behind this discrepancy is thought to be the muting effect held by averaging herd statistics on the extreme animals. In other words, representative animal approaches fail to sufficiently consider burdens arising from poorly performing animals, whose emissions intensity becomes exponentially (as opposed to linearly) higher as their ADG nears zero; this results in "empty" methanogenic emissions to merely sustain, rather than increase, their bodyweights. Indeed, the upper limit values of the 95% confidence intervals estimated by the Monte Carlo method were found to be smaller than the emissions intensities derived for the worst-performing "real" animals, and considerably so for the WC and HS systems under which ADG tended to be more variable **(Table 3.5)**.

Figure 3.8 depicts the above Monte Carlo results diagrammatically, alongside the uncertain ranges derived for the best and worst performing animals on each farmlet. Here, the hypothesis about the importance of considering the "weakest" animals seems to gain further credibility, as the worst performing animals demonstrate far wider 95% ranges than the average (and best performing) animals—especially for the WC system, whose overall burdens are more strongly affected by livestock emissions due to its lower ammonium nitrate (NH_4NO_3) application rates. While the ranges given for the best and worst performing animals may be slightly overestimated compared to those given for the average animals (as the animal heterogeneity has been considered twice, first in the form of animal performance and then through parameter distributions), this condition applies equally to both the best and worst animals; it would be reasonable to conclude, therefore, that the use of pre-averaged data *may* result in underestimation of emissions intensities, both in terms of point estimates as well as the width of confidence intervals. It should be noted, however, that this finding has only been drawn in the context of farm-scale LCA, and its relevance at the regional and national scales is not necessarily straightforward.

3.3.4 Sensitivity analysis

The results of the sensitivity analysis for allocation methods showed that the ratios derived for mass allocation of pasture-originating burdens did not deviate from the values originally prepared for economic allocation in any considerable manner. On average across the three systems, emissions assigned to cattle were 78% under economic allocation (i.e. 22% assigned to sheep) and 72% under mass allocation. This finding offers an interesting insight about the livestock market in the UK that the economic values of livestock products are strongly correlated with the amount of feed required to produce them. As a result, emissions intensity for WC, PP and HS, respectively, decreased by 2%, 3% and 4% under mass allocation of pasture, suggesting that modelling results were robust to the allocation method adopted.

Using the old IPCC conversion factors (IPCC, 2007) was found to affect model outputs, with both the mean emissions intensity for WC (14.3 kg CO_2-eq/kg LWG) and HS (18.9 kg CO_2-eq/kg LWG) resulting in significantly lower emissions intensity ($p < 0.001$ and $p = 0.027$). There was, however, no significant difference for the PP system, a result caused by lower CH_4 emissions from the system as a consequence of higher ADG. Since applying different conversion factors can significantly alter the results of a study, this adds another dimension to the already challenging cross-comparability issue in LCA research (McAuliffe et al., 2016). Future studies should be mindful that their estimate of emissions intensity may be higher than older work that employs the 2007 coefficients not because of the inefficiencies of farming strategies but because of the different assumptions adopted, particularly if the systems in question operate under low animal productivity.

3.4. Conclusion

This study used two approaches to calculate the partial carbon footprint of three pasture-based beef-cattle finishing systems trialled on the NWFP. In the first approach, emissions intensities were calculated for individual animals, whereas pre-averaged livestock data were utilised in the second approach. The results suggested that the outputs derived from pre-averaged data may be underestimated due to insufficient consideration given to uncertainty surrounding the herd structure and in particular poorly performing animals. The systematic bias identified by this study calls for careful interpretation of potentially optimistic LCA results based on pre-averaged herd data. At the same time, it opens up a large opportunity to reduce carbon footprints associated with livestock production systems, as the environmental benefit of evidence-based animal selection is likely to be considerably larger than currently thought and aligned with performance metrics i.e. ADG, which will encourage greater farm adoption.

Although animal heterogeneity was found to be a significant driving force behind livestock emissions intensities, the Monte Carlo analysis applied to individual animals simultaneously demonstrated that a large degree of uncertainty also exists around emission factors that have been assumed constant across treatments. This was particularly evident in PP and HS where considerably higher levels of N fertiliser were applied relative to WC (**Figure 3.8 and Table 3.1**). Given that these uncertainties are attributable to the current common practice of using recommended “book values” and national inventories regardless of the weather, soil, sward type, animal breed and management under which the farm operates, it can be hypothesised that replacing these values with site-specific emission factors will likely improve accuracy of farm-level assessments. As a prerequisite to test this hypothesis, Chapter 4 will derive original emission factors for pasture-originated N_2O emissions at each of the three farmlets at the NWFP.

Table 3.1. Inventory of material inputs for each system.

Variable	Unit	PP[d]	WC[e]	HS[f]
Area	ha	21.61	20.85	21.45
Fertiliser area	ha	21.24	20.52	21.03
FYM[a] area	ha	18.90	17.97	18.34
Yield	kg DM/ha	11474	10780	10417
Fertiliser applied				
N	kg	4951	681	4346
P	kg	206	1125	208
K	kg	554	454	312
Lime	kg	0	3002	4361
FYM[a]	t	118	118	98
Pesticides				
Glyphosate	kg	0	7.51	15.25
Fluroxypyr	kg	0	0	0.98
Seeds				
Grass	kg	0	734	650
Clover	kg	0	42	0
Diesel for machinery	l	342	1181	1295
Soybean	kg	651	651	672
Straw	kg	38920	39894	39685
Transport				
Soybean (sea)	tkm	6267	6267	6469
Soybean (road)	tkm	155	155	160
Straw (road)	tkm	2436	2497	2484
Fertiliser (road)	tkm	2444	2252	3949
Pasture quality				
DE[b]	%	77.55	77.7	76.78
CP[c]	%	20.72	20.12	17.41
Silage quality				
DE[b]	%	65.76	64.05	64.66
CP[c]	%	11.44	9.24	11.92

[a] FYM: farmyard manure
[b] DE: digestible energy
[c] CP: crude protein
[d] PP: permanent pasture
[e] WC: white clover/high sugar grass mix
[f] HS: high sugar grass monoculture

Table 3.2. Distributions of uncertainty parameters assumed in Monte Carlo simulations.

Emission source	Uncertainty	Distribution	Reference
Animal/housing			IPCC (2006)
Methane (EF[a] and MM[b])	± 20 %	Triangular	
Nitrous oxide (direct MM[b])	$SD^2 = 2$	Lognormal	
Nitrous oxide (indirect MM[b] leaching)	-1500%/333%	Triangular	
Nitrous oxide (indirect MM[b] volatilisation)	$SD^2 = 5$	Lognormal	
Pasture			IPCC (2006)
Nitrous oxide (direct)	$SD^2 = 3$	Lognormal	
Nitrous oxide (indirect leaching)	-1500%/333%	Triangular	
Nitrous oxide (indirect volatilisation)	$SD^2 = 5$	Lognormal	
Carbon dioxide (lime)	-50%/0%	Triangular	

[a] Enteric fermentation; [b] Manure management

Table 3.3. Livestock performance under each system.

Parameter	Unit	PP[a]	(SD[b])	WC[c]	(SD)	HS[d]	(SD)	*p*-value[e]
Entry weight	kg	279	(32.08)	279	(28.76)	284	(35.76)	0.80
Finishing weight	kg	607	(50.75)	582	(47.15)	590	(39.23)	0.12
Total growth	kg	328	(41.68)	304	(45.73)	307	(38.67)	0.05
Time on Farm Platform	d	448	(40.33)	461	(43.68)	453	(31.97)	0.46
Average daily weight gain	kg/d	0.76	(0.10)	0.68	(0.10)	0.70	(0.08)	< 0.01

[a] PP: permanent pasture
[b] SD: standard deviation
[c] WC: white clover/high sugar grass mix
[d] HS: high sugar grass monoculture
[e] Based on multi-sample F-tests

Table 3.4. Factors contributing to emissions intensity of individual cattle. Results are presented as the average value across 30 cattle assigned to each system in the unit of kg CO_2-eq/kg LWG.

Source[a]	PP[b]	(Range)	WC[c]	(Range)	HS[d]	(Range)
Enteric fermentation (CH_4)	7.09	(6.16 - 8.02)	7.7	(6.43 - 9.70)	7.52	(5.24 - 9.61)
Manure management (CH_4)[e]	1.36	(0.73 - 1.78)	1.83	(1.11 - 2.56)	1.68	(1.27 - 2.59)
Manure management (direct N_2O)	1.15	(0.99 - 1.34)	1.06	(0.66 - 1.38)	1.06	(0.70 - 1.32)
Manure management (indirect volatilisation N_2O)	0.2	(0.17 - 0.22)	0.18	(0.11 - 0.23)	0.18	(0.12 - 0.23)
Barley production	0.56	(0.44 - 0.69)	0.62	(0.46 - 0.90)	0.61	(0.50 - 0.87)
Ammonium nitrate production	3.56	(2.78 - 4.39)	0.53	(0.39 - 0.76)	3.32	(2.73 - 4.72)
Fertililser application (N_2O)	2.03	(1.59 - 2.50)	0.3	(0.23 - 0.44)	1.89	(1.56 - 2.69)
Urine and dung from ewes on pasture (N_2O)	0.6	(0.47 - 0.74)	0.67	(0.50 - 0.97)	0.66	(0.54 - 0.94)
Farmyard manure application (N_2O)	0.43	(0.25 - 0.55)	0.45	(0.30 - 0.62)	0.52	(0.40 - 0.78)
Crop residues (N_2O)	-	-	0.33	(0.25 - 0.48)	0.33	(0.27 - 0.46)
Indirect emissions from leaching (N_2O)	0.2	(0.17 - 0.24)	0.11	(0.08 - 0.14)	0.2	(0.15 - 0.29)
Urine and dung from cattle on pasture (N_2O)	0.25	(0.19 - 0.31)	0.21	(0.13 - 0.29)	0.19	(0.11 - 0.27)
Single superphosphate production	0.03	(0.02 - 0.04)	0.18	(0.14 - 0.27)	0.03	(0.03 - 0.05)
Others[f]	1.03	(0.80 - 1.26)	1.8	(1.34 - 2.58)	1.97	(1.61 - 2.80)
Total	18.47	(16.32 - 21.71)	15.96	(13.73 - 20.90)	20.17	(16.63 - 25.61)

[a] Results are presented as the average value across 30 cattle assigned to each system in the unit of kg CO2-eq/kg LWG.

[b] PP: permanent pasture

[c] WC: white clover/high sugar grass mix

[d] HS: high sugar grass monoculture

[e] Methane arising from manure management was calculated under a deep bedding system assuming a methane conversion factor of 20% and an average annual temperature of 12°C.

[f] Includes processes which account for < 1% of the total emissions intensity: lime production and decomposition, soybean production, pesticide production, transportation and diesel combustion for machinery.

Table 3.5. Comparison of emissions intensity (kg CO_2-eq/kg LWG) derived under two methods.

System	Representative animal approach				Individual animal approach			
	Mean	LL[a]	UL[a]	Range	Mean	Min	Max	Range
PP	17.8	15.0	21.5	6.5	18.4	16.3	21.7	5.4
WC	14.4	12.7	16.2	3.5	16.0	13.7	20.9	7.2
HS	19.0	16.3	22.5	6.2	20.2	16.6	25.6	9.0

[a] Lower and upper limit values of the 95% confidence interval estimated by Monte Carlo simulations.

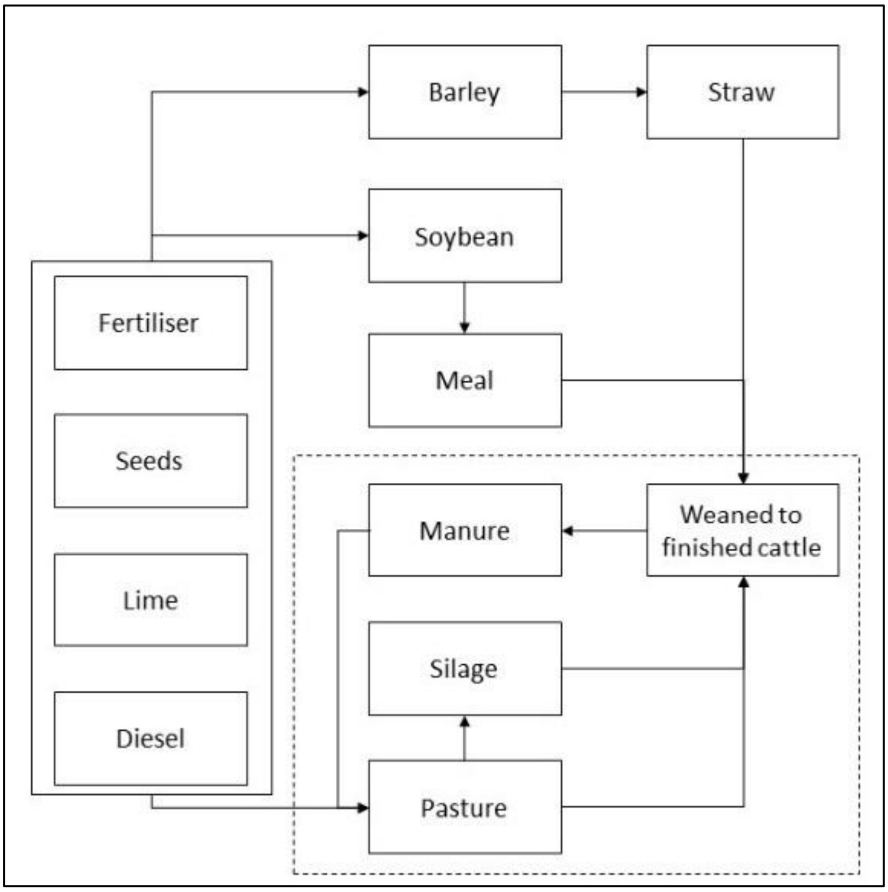


Figure 3.1. System boundary of the present study. The dashed line represents the North Wyke Farm Platform.

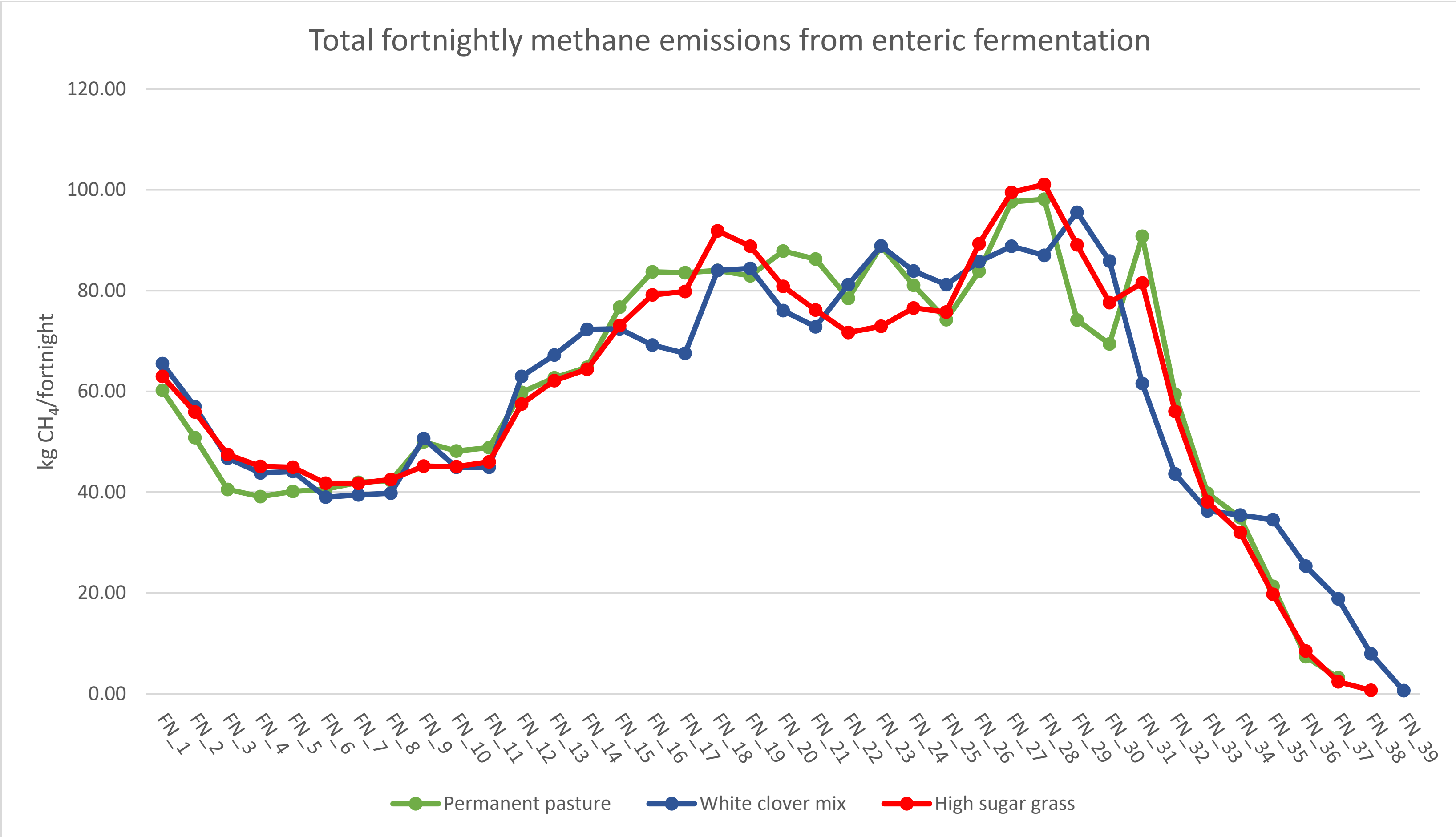


Figure 3.2. Temporal variations in methane emissions from enteric fermentation. The values are aggregated across 30 cattle under each system. FN = fortnight.

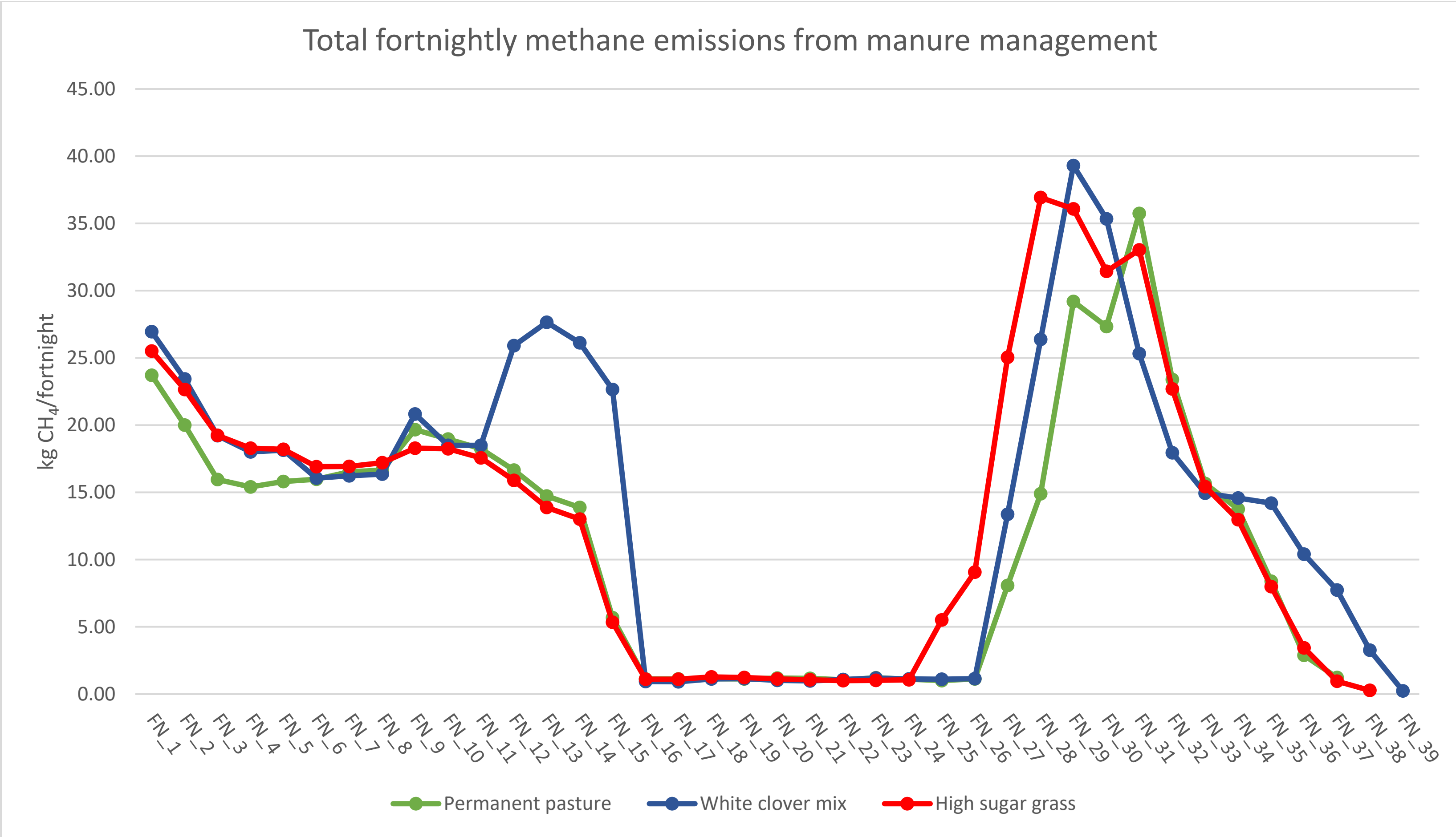


Figure 3.3. Temporal variations in methane emissions from manure management. The values are aggregated across 30 cattle under each system. FN = fortnight

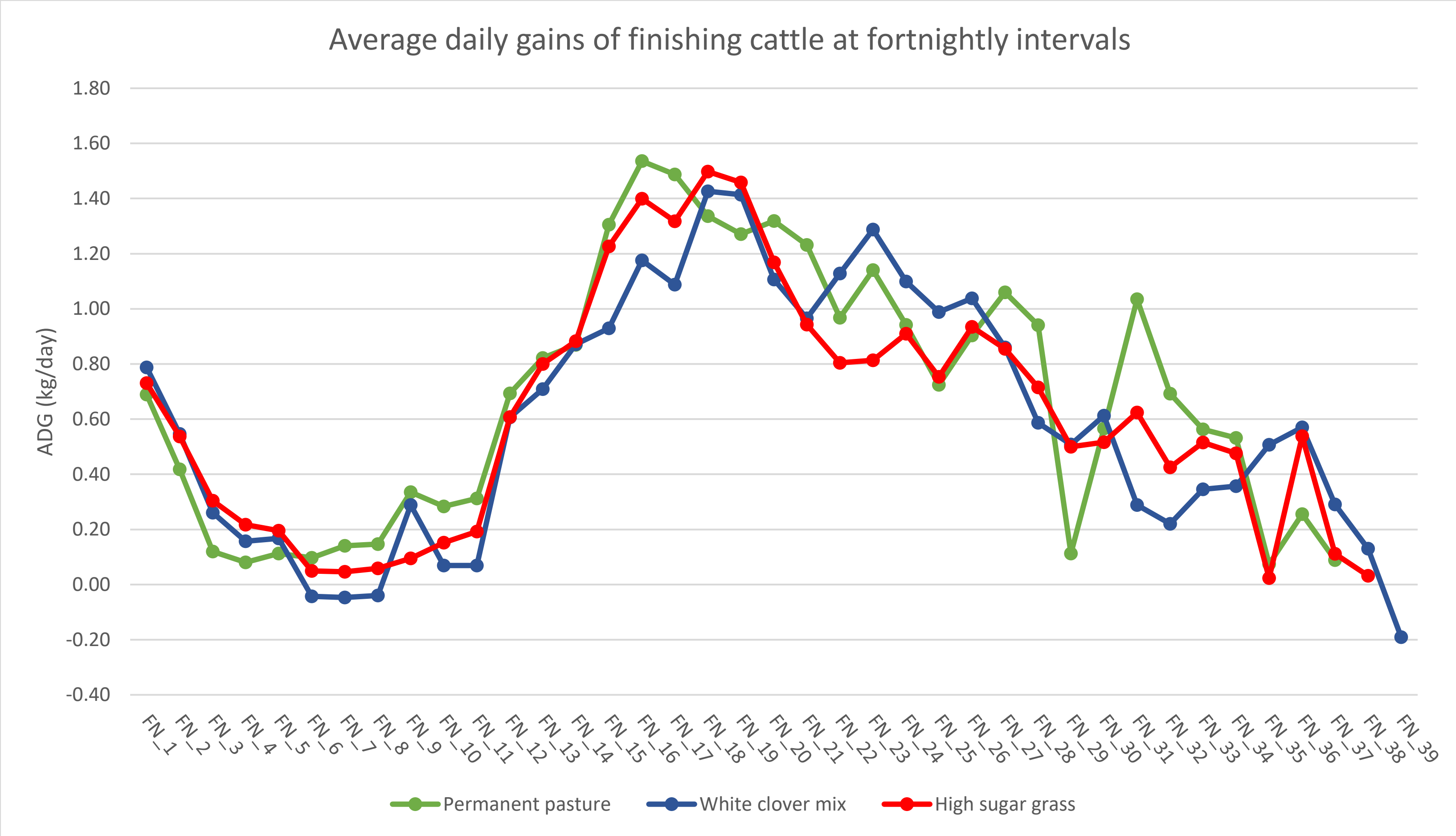


Figure 3.4. Temporal variations in average daily gains (ADG). FN = fortnight.

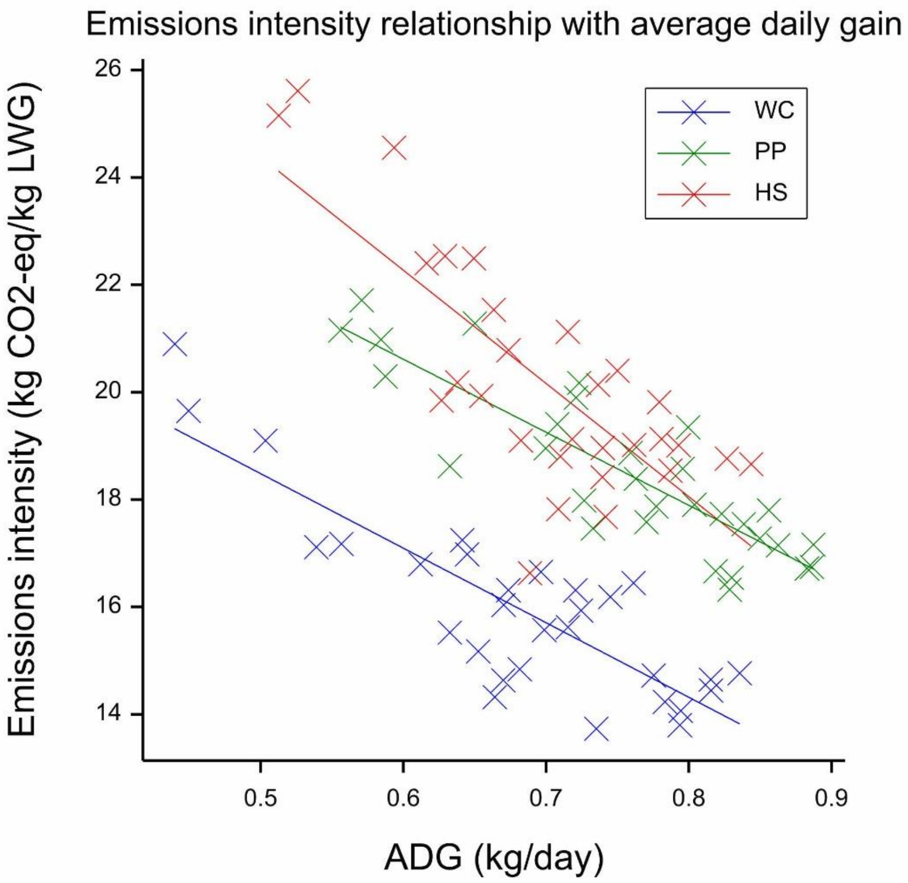


Figure 3.5. Relationship between emissions intensity and average daily gains (ADG) under each system. PP: permanent pasture; WC: white clover/high sugar grass mix; HS: high sugar grass monoculture.

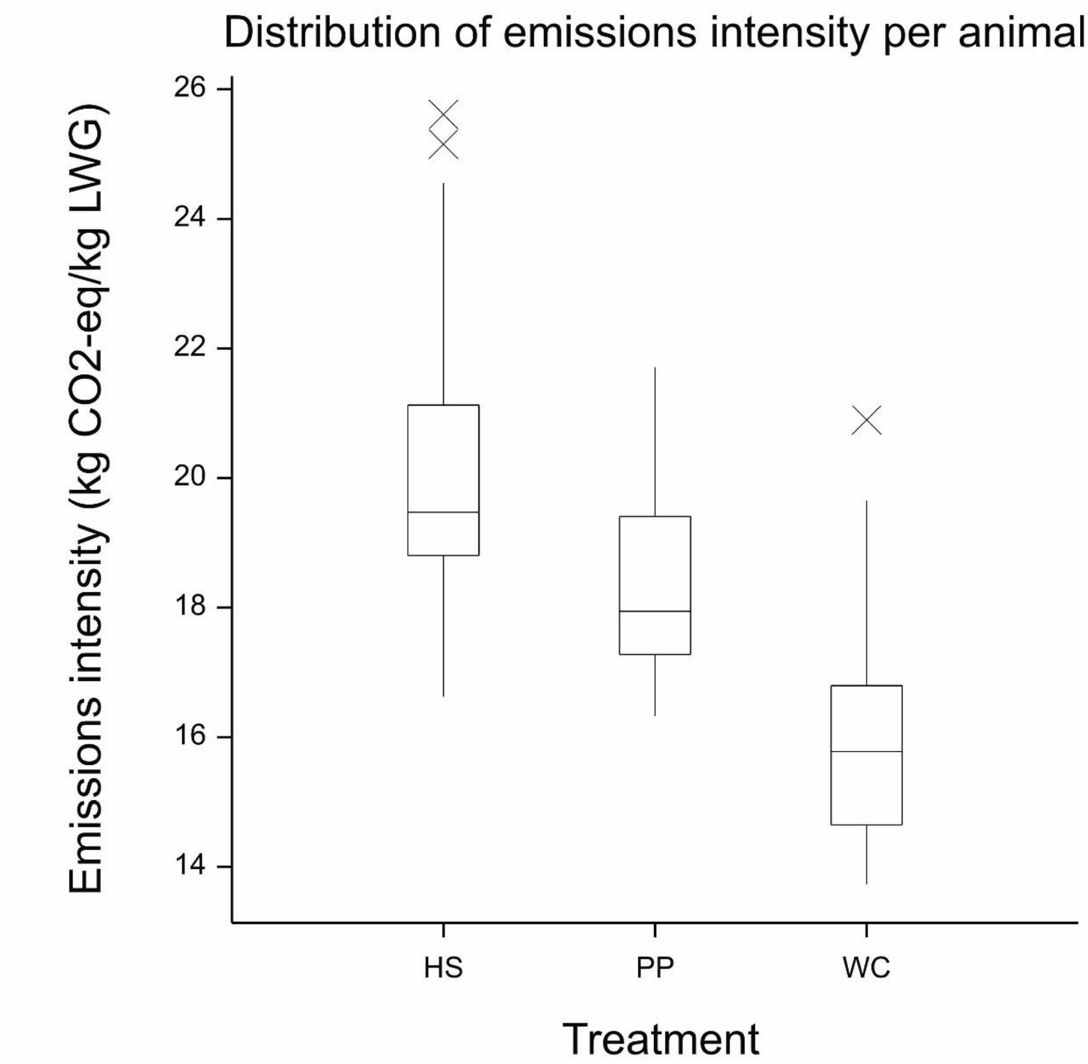


Figure 3.6. Distribution of emissions intensity per animal under each system. Outliers located further than 1.5 times the interquartile range beyond the quartiles are each denoted with a cross (×).

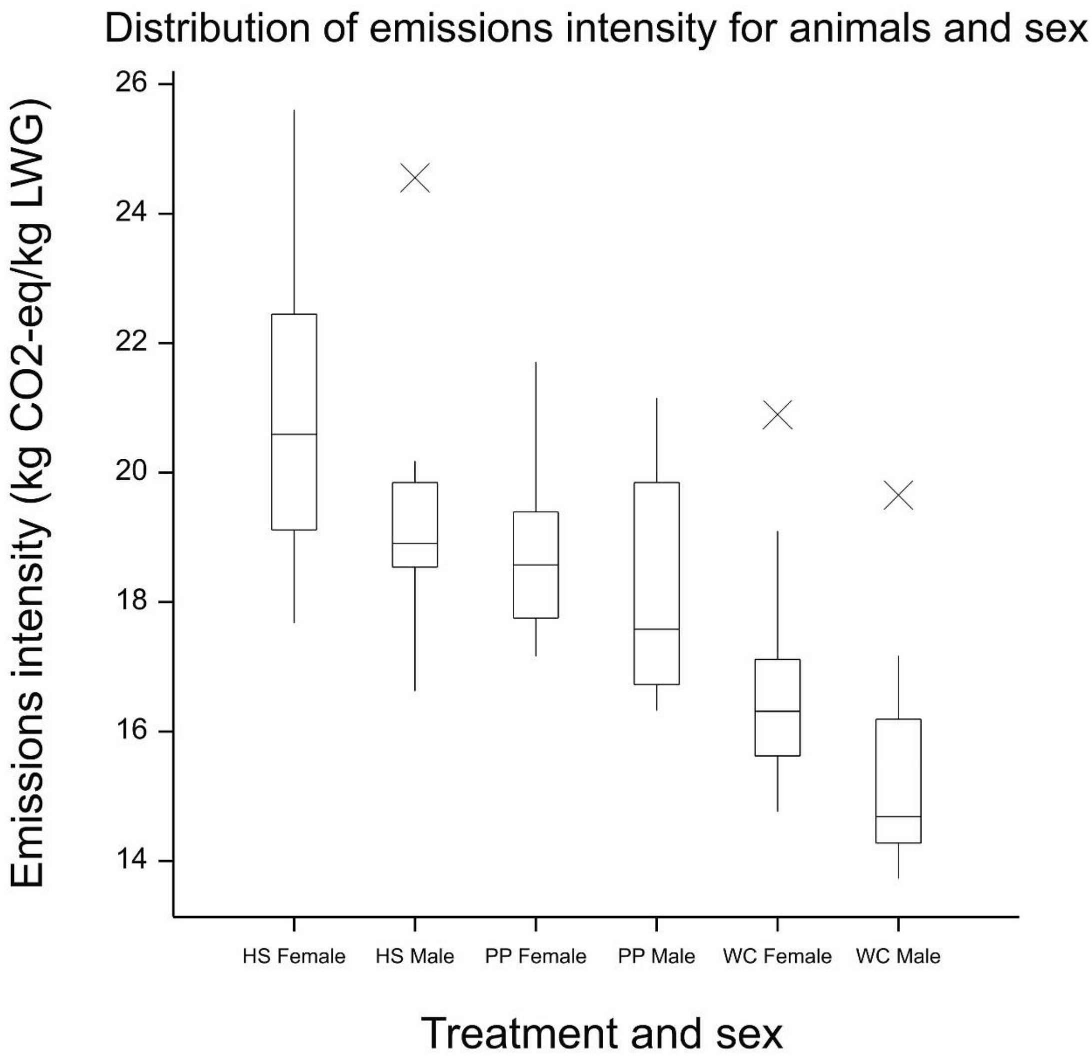


Figure 3.7. Distribution of emissions intensity per animal by sex. Outliers located further than 1.5 times the interquartile range beyond the quartiles are each denoted with a cross (×).

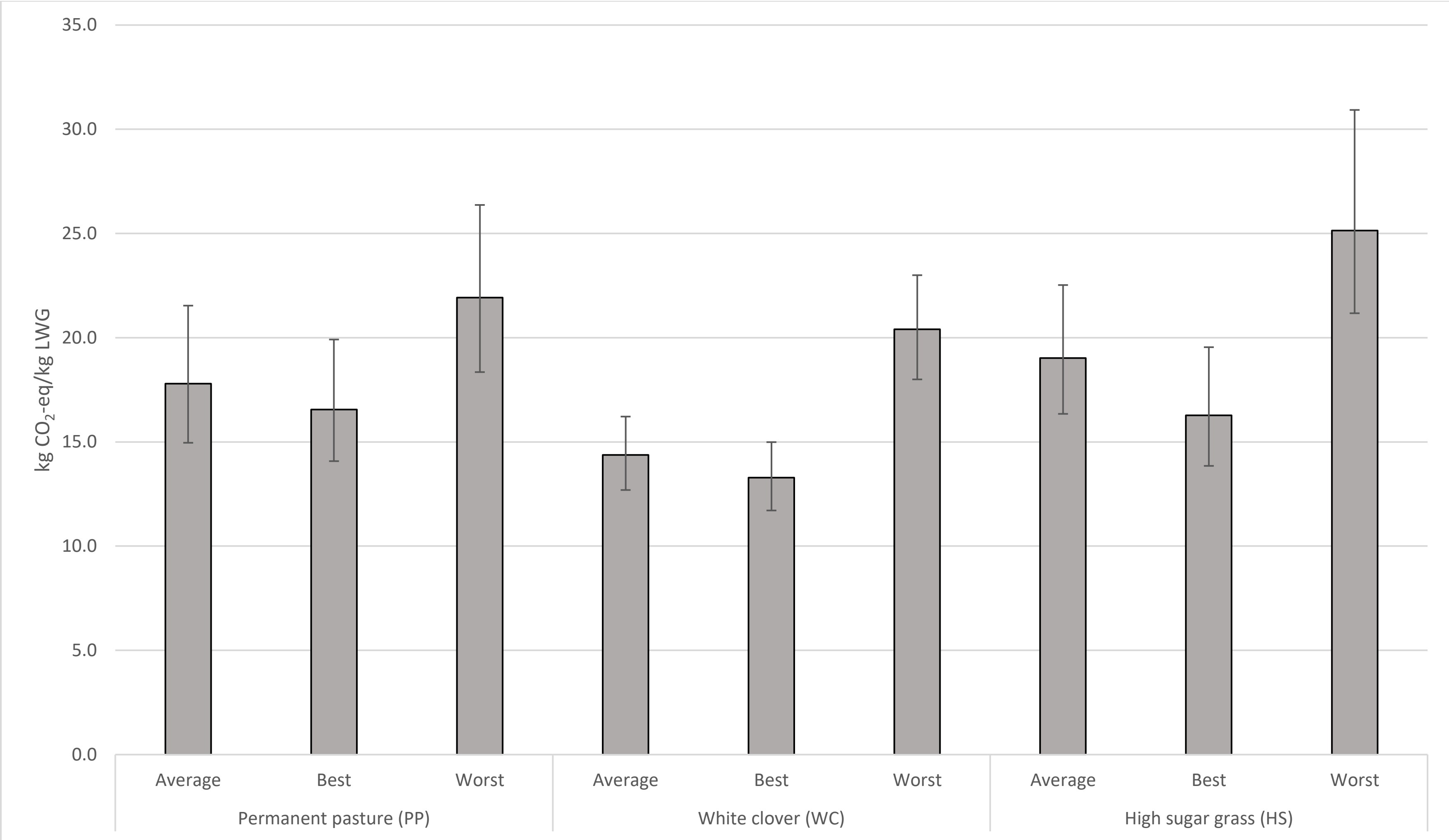


Figure 3.8. Results of Monte Carlo simulations applied to pre-averaged representative animals and the best and worst performing animals. Error bars represent 95% confidence intervals.

Chapter 4 – Deriving site-specific emission factors to reduce uncertainties inherent in life cycle assessment

4.1 Introduction

The majority of carbon footprint (CF) studies in the agricultural sector—including those carried out in Chapters 2 and 3 of this thesis—employ pre-defined emissions factors (EFs), or fixed parameters linking nutrient input with greenhouse gas (GHG) output from the system, as part of the estimation process. As the actual ratio between the two values varies depending on the production environment of the farm, such as: climate, soil, plant and animal genetics as well as management practice, a considerable level of uncertainty surrounds these universal EFs. For example, the two parameters for nitrous oxide (N_2O) emissions supplied by IPCC (2006), commonly known as EF_1 (% fertiliser N lost as N_2O) and $EF_{3(PRP)}$ (% urine and dung N deposited on pasture lost as N_2O, henceforth referred to as EF_3), respectively, are deemed to have a 95% confidence interval between 67% and +300% of the point estimates. This, in turn, generally leads to less accurate estimates of N_2O losses compared to locally conducted field-scale experiments (Misselbrook et al., 2014). Given that EF_1 and EF_3 are only a subset of EFs required to calculate the agricultural carbon footprint (CF), this demonstrates the importance of reducing uncertainty inherent in EFs at both local and national scales.

Motivated by this observation, the current chapter reports results of an original field trial measuring N_2O fluxes at the three farmlets of the NWFP (Section 1.4), with a view to derive site-specific EFs that can be fed into an LCA study to reduce its uncertainty (Chapter 5). Agriculture is one of the greatest net contributors of N_2O (Galloway et al., 2004, Reay et al., 2012), a GHG ~265 times more potent than carbon dioxide (CO_2) (IPCC, 2013) with an atmospheric residence time of ~116 years (Prather et al., 2015). Globally, the majority of N_2O emissions arising from agricultural production occur due to manure and faeces left on pasture (36%) and the application of synthetic fertilisers (28%) (FAO, 2016). While the UK achieved a reduction of 52% in total N_2O emissions over the period between 1990 and 2013, the agricultural sector only managed a 16% reduction and, as a result, associated soils account for as much as 70% of the country's total N_2O emissions (DECC, 2015). The importance of reducing soil-originated N_2O emissions was also evident in Chapter 3, where they were shown to account for 26% of life cycle GHG emissions attributable to the NWFP's cattle finishing (from weaning) enterprise.

In addition to the primary objective of supplying localised EFs to the next chapter, the current chapter also offers a novel contribution to the global warming literature. As mentioned above, N_2O measurements reported here were taken under three distinct pasture management strategies within the NWFP, which represent the three most common pastures within the UK (permanent pasture (PP); *Lolium spp.*); white clover (WC; *Trifolium repens*) leys; and high sugar grass (HS) monocultures. Although studies have considered the effect of land use change (LUC) on N_2O emissions from soil, for example when pasture was converted to cropland (Reinsch et al., 2018), no identified studies have investigated differences between multiple sward types under similar soil conditions. Doing so will enable the separation of the weather and soil effects from management effects; with the NWFP including both permanent and short-term ley pastures, climate change implications of reseeding can also be quantified.

4.2 Materials and methods

4.2.1 Study site

The field trial was conducted across the three farmlets of the NWFP (Section 1.2). One grazing field per farmlet was selected based on similarities with regards to the field size and animal rotation patterns. These fields were utilised extensively by beef finishing-cattle over the grazing season (April 2017 to October 2017) in which this work was carried out. Basic information on soil characteristics for each of the fields are provided in **Table 4.1**.

Figure 4.1 displays 30-year means for rainfall and temperature at North Wyke. Rainfall is generally highest in December (130 mm) while temperature tends to peak in July (max. = 19.9°C; min. = 12.0°C) and August (max. = 19.8°C; min. = 12.1°C). Lowest values are observed during June for rainfall (55.9 mm), and January (max. = 7.7°C; min. = 2.5°C) and February (max. = 7.7°C; min. = 2.1°C) for temperature.

4.2.2 Experimental design

The experiment proper, defined as the duration of gas sampling including what took place prior to the application of treatments, commenced on 11/04/2017 and concluded on 27/09/2017. On each of the three fields, three experimental blocks (15 m x 1.5 m) were established at locations approximately equidistant from the centre of the field. These blocks were fenced off using electric wiring while animals were grazing on the same field, but otherwise managed similarly to the rest of the pasture with grazing simulated through grass-mowing. Each block was further divided into six plots (2.5 m x 1.5 m) laid along a contour, and randomly assigned to either a treatment or a control. As only four of these plots were required to derive emission factors, purposes of and measurements from the other two will not be reported here and fulfilled other scientific investigations. Within each plot, a 1 m^2 area was set aside for gas measurements, and two static chambers (0.16 m^2 each) were tightly installed ~5cm into the soil within this space. Treatments were defined as dung (D) and urine (U). Two types of control plots were also established: the first group received inorganic N at the same rate as the treatment plots and the rest of the field (CON+N), while another received no N of any form (CON-N). The latter control was required to derive the EF for synthetic fertilisers, or EF_1 as defined by IPCC (2006).

Urine and dung were applied at a rate of 5 l/m^2 (de Klein et al., 2014) and 20 kg/m^2 (Cardenas et al., 2016), respectively. These values represent approximate rates typically returned during a single deposition event. Urine was collected from cattle grazing each farmlet after segregating steers and heifers. Samples from steers were obtained via an improvised bucket with an extended handle during natural urination events. Heifers, on the other hand, were put through a cattle crush and encouraged to urinate using vulva stimulation. Following on-field collection, urine was frozen at – 20°C for each animal until the day before application, when it was defrosted and mixed together into bulked barrels for PP, WC and HS separately. Dung was collected from respective fields with a barrel and ladle through subjective identification of the freshest dung pats. Following collection, samples were homogenised and refrigerated at 4°C until the day of application. On 13/06/2017 urine was applied to U plots. Application to the areas inside and outside chambers was carried out separately using watering cans with perforated spray heads. Dung was applied evenly, again separately to inside and outside chambers, to D plots at the aforementioned rate. Urine samples were analysed using a Shimadzu TOC / TN analyser, with total N content (as nitrogen oxides) determined by chemiluminescence after catalytic thermal decomposition at 720°C. Total N content of dung applied was quantified following the same method as that for herbage samples described in Section 3.2.2.

Following the NWFP's standard farm management practices, inorganic N fertiliser (ammonium nitrate, NH_4NO_3) was applied to PP and HS (except for CON-N plots), three times during the grazing season on 10/04/2017, 08/05/2017 and 05/06/2017, at a rate of 40 kg N/ha per application event. To ensure areas inside chambers received the correct amount of fertiliser, all chambers were closed during the field-wide application by spreader and subsequently received the same fertiliser at the same rate by hand. CON-N plots were completely covered throughout this process and received no form of N. Unlike PP and HS systems, white clover mixed sward (WC) plots did not receive any inorganic N. Consequently, two CON-N plots (and no CON+N plot) were created on each WC block. Throughout the course of the experiment proper, no other forms of fertiliser (e.g. phosphorus or potassium) were applied to any of the fields.

4.2.3 Sampling

Gas sampling and analysis was carried out according to the experimental protocol developed by Chadwick et al. (2014). To accurately estimate EFs representative of the entire grazing season, sampling commenced on 11/04/2017 (63 days before treatment application) immediately following the first inorganic N application. It concluded on 27/09/2017 (106 days after treatment application) when emissions from all plots had come back to pre-season levels.

Prior to treatment application, N_2O sampling was conducted three times a week for a fortnight following each application of inorganic fertiliser, and twice weekly thereafter. After urine and dung were applied, gases were again collected three times a week for the first two weeks and then twice weekly for the next 11 weeks. From 13/09/2017 (92 days after treatment application) onwards, the frequency was reduced to once fortnightly. Overall, this resulted in 43 individual sampling days over a 169-day period. On each sampling day, chambers were closed at 11am (T0) after ambient gas samples were collected. Gases from inside chambers were then sampled 40 minutes later (T40). In addition, one chamber per block was designated as a linearity chamber (Chadwick et al., 2014), where samples were also collected at 20 minutes (T20) and 60 minutes (T60) to test the assumption of temporally linear gas accumulation within chambers. Once samples were collected in sealed glass vials, N_2O flux was estimated using a Perkin Elmer Clarus 500 gas chromatograph, which was fitted with a Turbomatrix 110 automated headspace sampler and an electron capture detector set at 300°C. Separation was achieved by a Perkin Elmer Elite-PLOT megabore capillary column (30 m long and 0.53 mm i.d.) maintained at 35°C, with nitrogen (N_2) used as the carrier gas (Cardenas et al., 2016). The obtained values were adjusted for the soil temperature recorded at each sampling event. Finally, using the flux values derived for all sampling days, cumulative emissions to represent the entire grazing season were calculated using trapezoidal integration (Cardenas et al., 2010).

4.2.3.2 Soil moisture

On each gas sampling day post-treatment application, soil moisture at each block was measured using the gravimetric method. Soil samples were only taken from CON-N and D plots for this purpose, as the results of multiple spot sampling confirmed that no statistically significant difference was observed amongst plots other than D within a single block. Soil moisture values were subsequently converted to water filled pore space (WFPS), using bulk density values identified separately for each plot, to account for the degree of soil compaction. Soil temperature at 5 cm was also measured at every gas sampling event on all plots using a portable thermometer (Fisher Scientific, UK).

4.2.4 Data analysis

Statistical analysis was conducted using GenStat V18 (www.vsni.co.uk/software/genstat). Cumulative emissions were log-transformed to account for skewed residual distributions and subsequently analysed using restricted maximum likelihood models (REML) with a block structure of field block/plot/chamber. REML was chosen over analysis of variance (ANOVA) due to the uneven design of the experiment, resulting from the fact that there are two CON-N plots on WC and one on PP and HS. Effects of treatments and farmlets as well as their interactions were examined. Differences in WFPS across farmlets and between CON-N and D plots were tested using two-way ANOVA. To investigate the relationship between WFPS and N_2O emissions, a simple linear regression was also carried out.

EF_1 was calculated by subtracting the cumulative emissions on CON-N plots from those on CON+N plots within the same block and dividing by the amount of N applied to CON+N plots **(Table 3)**. EFs for urine (*EF_U*) and dung (*EF_D*), on the other hand, were estimated by subtracting emissions on CON+N plots from those on U plots and D plots, respectively, and dividing by the amount of N applied in the form of excreta. Finally, to obtain a single *EF_3* value representative of manure, the weighted average between *EF_U* and *EF_D* was calculated using the commonly adopted ratio of 60:40 (Cardenas et al., 2010). As discussed, these values will be used in the LCA study carried out in Chapter 5.

4.3 Results and discussion

4.3.1 Weather

Rainfall and temperatures recorded during the experiment proper are depicted in **Figure 4.2**. In comparison to the 30-year average **(Figure 4.1)**, rainfall was considerably higher throughout the experiment, with the highest monthly total occurring in July (105 mm). Total rainfall in the 30 days following treatment application was 55 mm, with the distribution skewed towards the end of June after a comparatively dry period. From treatment application to the conclusion of the trial, 296 mm of precipitation was recorded. In contrast to rainfall, air temperature followed a similar pattern to the 30-year average, although minimum monthly averages were slightly higher in the year of the experiment. The highest maximum daily temperature recorded was 29°C on 22/06/2017, whereas the lowest minimum was -2°C on 26/04/2017. On the day of treatment application, the temperature was 14.4°C at maximum and 8.9°C at minimum. The average maximum and minimum temperatures for the 60 days following the treatment application were 19.8°C and 12.1°C, respectively.

4.3.2 Nitrogen application rates

Between two treatments (U and D, from their respective cattle), N input per area (representing three inorganic N applications and a single event of deposition) was higher for D plots on all farmlets, with HS recording the highest values (911 kg/ha) followed by PP (784 kg/ha) and then WC (559 kg/ha) **(Table 4.2)**. U plots received relatively low levels of N, with the most applied to PP (286 kg/ha), followed by HS (207 kg/ha) and then WC (98 kg/ha). CON+N plots, by design, received 120 kg N/ha over the experiment proper, while CON-N plots, also by design, did not receive any N. Across all plots (other than CON-N), WC had notably lower N inputs than the other two systems, due largely to the absence of mineral fertiliser N and the complete reliance on biological N fixation from the legume, which would be more variable.

4.3.3 Nitrous oxide emissions

In total, 125, 129 and 122 sets of three samples (T20, T40 and T60) were tested for linearity on PP, WC and HS, respectively, with the discrepancies resulting from sampling errors or damaged chambers which resulted in samples being irretrievably lost on a given day. Amongst them, 88% of WC samples and 85% of both PP and HS samples were represented by a linear equation with $R^2 > 0.5$. These percentages are largely comparable to those reported by Cardenas et al. (2016).

Following treatment application on 13/06/2017, there were modest spikes on the treated plots **(Figure 4.3)**; however, the largest fluxes did not occur until the end of July, when the temperature and rainfall both became relatively higher. Both types of controls exhibited comparatively lower fluxes, with CON+N generating relatively larger peaks than CON-N (on PP and HS that received inorganic fertiliser). Fluxes were small at all control plots on WC, none of which received N fertiliser. Across all farmlets, D plots produced the largest fluxes, with HS generating a notably high daily peak (612 g N_2O/ha/d), followed by PP (236 g N_2O/ha/d) and WC (159 g N_2O/ha/d) **(Figure 4.3)**. U plots produced considerably lower N_2O emissions in comparison to D plots; the maximum daily fluxes for U were 115, 61 and 65 g N_2O/ha/d for PP, WC and HS, respectively.

Total emissions across the experiment proper are summarised in **Table 4.3**. The results of REML estimation revealed that these values were significantly different amongst treatments ($p < 0.001$) but not farmlet ($p = 0.102$). A significant interaction between treatment × farmlet was also observed ($p < 0.001$). CON+N tended to have considerably higher emissions than CON-N. PP was found to produce more N_2O than HS on CON+N plots, while HS generated the highest N_2O emissions on CON-N, U and D plots. WC was the lowest contributor of N_2O on U and D plots but not on CON-N plots, where its emissions were higher than PP.

Contrasts can be drawn from two earlier GHG measurements carried out at the NWFP. Cardenas et al. (2016) recorded cumulative N_2O values of 3,192 and 3,244 g N/ha for U and D, respectively, under a summer application of treatments on a permanent pasture. With urine and dung brought in from separate herds, N input on the U treatment was 429 kg N/ha, a level considerably higher than what was applied under the current study. Emissions from U plots were lower and D plots were higher in the present work, although care should be taken with such comparisons as the duration of sampling campaigns do not accurately coincide. In a grassland monitoring study without treatment application, Horrocks et al. (2014) recorded a daily maximum flux of under 47.3 g N_2O/ha/d on intensively managed plots, which is considerably lower than 155 g N_2O/ha/d recorded on PP CON+N, the most comparative group in the current study. It should be noted, however, that amount of fertiliser applied in the 2014 study (80 kg N/ha) was also lower than the present campaign.

4.3.4 Soil moisture

The maximum level of soil moisture was recorded on the HS farmlet in the middle of August **(Figure 4.4)**. Throughout the season, however, PP generally maintained the highest moisture levels. Across three farmlets and the two groups on which moisture was measured (CON-N and D), there were significant effects of farmlets ($p < 0.001$) and the farmlet × treatment interaction ($p = 0.01$), whereas the treatment effect was relatively weaker (p = 0.09). Within each farmlet, moisture on D plots was higher than CON-N plots on PP ($p = 0.004$), while CON-N was higher than D on WC ($p = 0.05$). No statistically significant differences between CON-N and D were observed on HS ($p = 0.07$). Although a positive association was identified between soil moisture and N_2O emissions, this was not found to be significant either ($p = 0.25$).

Following treatment application, soil moisture (as expressed in WFPS) was in the range of 44% to 88%, with average values of ~60% between both treatments **(Figure 4.4)**. As already mentioned, there was a comparatively dry period with low rainfall directly after treatment application **(Figure 4.2)**; after 15.8 mm of rainfall on the night before treatment application, the following 21 days had minimal precipitation, coupled with generally high maximum daily temperatures. This led to relatively low soil moisture, a likely cause of the low N_2O responses during the first four weeks following treatment application. When major N_2O responses occurred in August, soil moisture was relatively high, generating conditions which favoured denitrification (AHDB, 2016).

4.3.5 Emission factors

Due to differences in levels of N input, the relative rankings for cumulative emissions did not entirely translate across to emission factors **(Table 4.4)**; for example, based on levels of N applied to levels of N_2O-N lost, PP (0.16 %) performed more favourably than WC (0.64%) for EF_3. HS remained the highest ranked (most polluting) for EF_U, EF_D and ultimately EF_3 (0.89%), while PP (1.38%) recorded a higher EF_1 than WC (0.73%). On average across farmlets, EF_3 calculated in the present study was 0.56%, considerably lower than the default IPCC (2006) EF_3 value (2%). Possible causes for these differences will be discussed in Chapter 5 along with the results of carbon footprint conducted under these EFs.

Previous research has argued that disaggregating EFs for manure into those for urine and dung is warranted (van der Weerden et al., 2011). The results of the present study suggest, however, that EF_U and EF_D do not vary very much, particularly on PP and WC **(Table 4.4)**. Furthermore, EFs estimated here were generally lower than those previously observed on the NWFP, for example of EF_U of 2.96% under spring application (Cardenas et al., 2016). These discrepancies may partly be explained by the low N values identified in cattle urine within the present study **(Table 4.2)**, which are known to reduce N_2O emissions per an amount of N input (Hoogendoorn et al., 2016).

4.4 Conclusion

To obtain site-specific EFs representative of weather, soil, plant and animal genetics as well as from farm management strategies at the NWFP, N_2O emissions were measured under two treatments (U and D) and two controls (CON+N and CON-N) on each of its three farmlets. Overall, HS plots tended to have higher cumulative emissions than PP and WC. EFs developed in the current study were found to be lower than those reported in previous research, possibly explained by the generally low N content of urine sampled from the NWFP cattle, demonstrating the importance of considering site-specific parameters in livestock LCA. The derived EFs will be incorporated into the carbon footprint framework in the next chapter, to reduce uncertainty associated with "generic" emission factors intended for the widest possible audience.

Table 4.1. Soil characteristics of each field used in the experiment.

Farmlet	Permanent pasture (PP)	White clover mix (WC)	High sugar grass (HS)
Field name	Orchard Dean	Higher Wyke Moor	Poor Field
Field size (ha)	3.92	4.32	3.92
Soil type	Clay	Clay	Clay
pH	5.64	5.47	5.74
Olsen P extractable (mg/l)	18.8	13.6	14.8
K extractable (mg/l)	213	145	207
Mg extractable (mg/l)	112	70	71
Organic C (%)	1.66	3.49	4.43
Total N (%)	0.62	0.40	0.41
Bulk density (g/cm^3)	0.88	0.98	1.08

Table 4.2. N inputs to each treatment (kg N/ha).

Farmlet	D[a]			U[b]			CON+N[c]	
	Fertiliser	Dung	Total	Fertiliser	Urine	Total	Fertiliser	Total
PP[d]	120	664	784	120	166	286	120	120
WC[e]	-	559	559	-	98	98	-	-
HS[f]	120	791	911	120	87	207	120	120

[a] Dung; [b] Urine; [c] Control with synthetic N.

[d] Permanent pasture; [e] white clover mix; [f] high sugar grass monoculture.

Table 4.3. Cumulative N_2O-N emissions (as g N_2O-N/ha).

Treatment	CON+N[a]	CON-N[b]	U[c]	D[d]
PP[e]	2320	670	2249	5380
WC[f]	NA	588	1176	4049
HS[g]	1893	1022	2597	9935

All values are averages across blocks and chambers

[a] Control with N; [b] Control without N; [c] Urine; [d] Dung

[e] Permanent pasture; [f] white clover mix; [g] high sugar grass monoculture.

Table 4.4. Derived emission factors. All values reported as % N lost as N_2O-N.

	PP[a]	WC[b]	HS[c]
EF_U[e]	-0.04	0.69	0.81
EF_D[f]	0.46	0.64	1.02
EF_3[g]	0.16	0.64	0.89
EF_1[h]	1.38	NA	0.73

[a] Permanent pasture; [b] white clover mix; [c] high sugar grass monoculture.
[e] Emission factor for urine; [f] Emission factor for dung; [g] Emission factor 3 for excreta deposited on pasture; [h] Emission factor for nitrogen fertiliser lost as nitrous oxide.

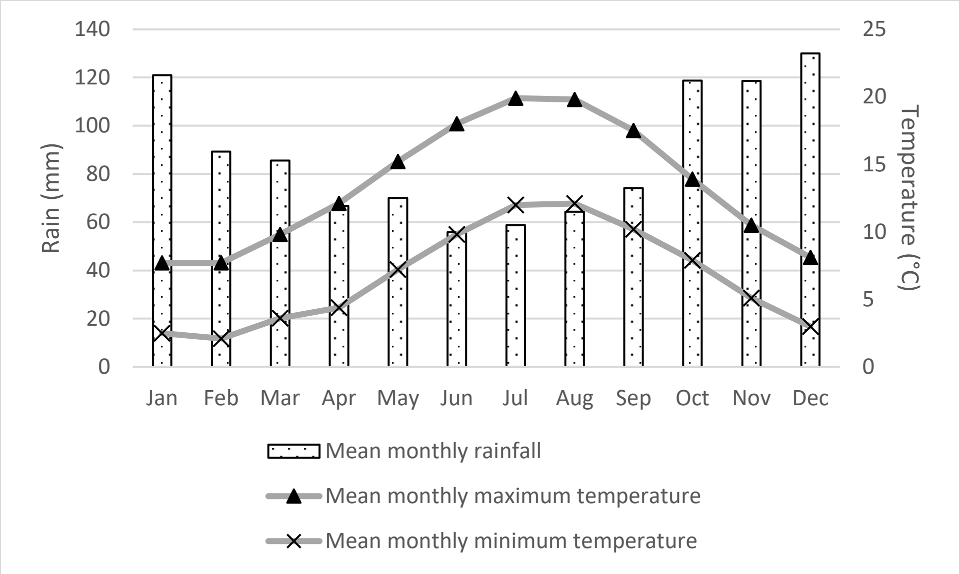


Figure 4.1. Mean monthly meteorological data for North Wyke between 1981 and 2010.

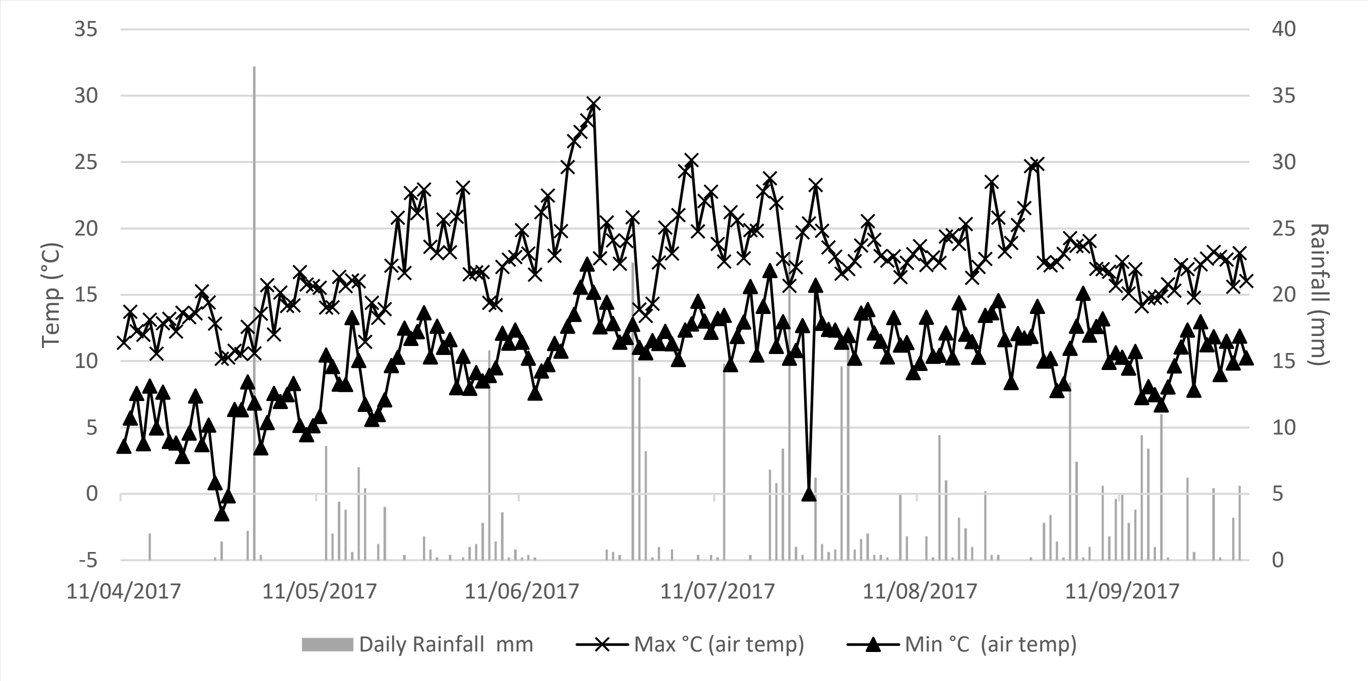
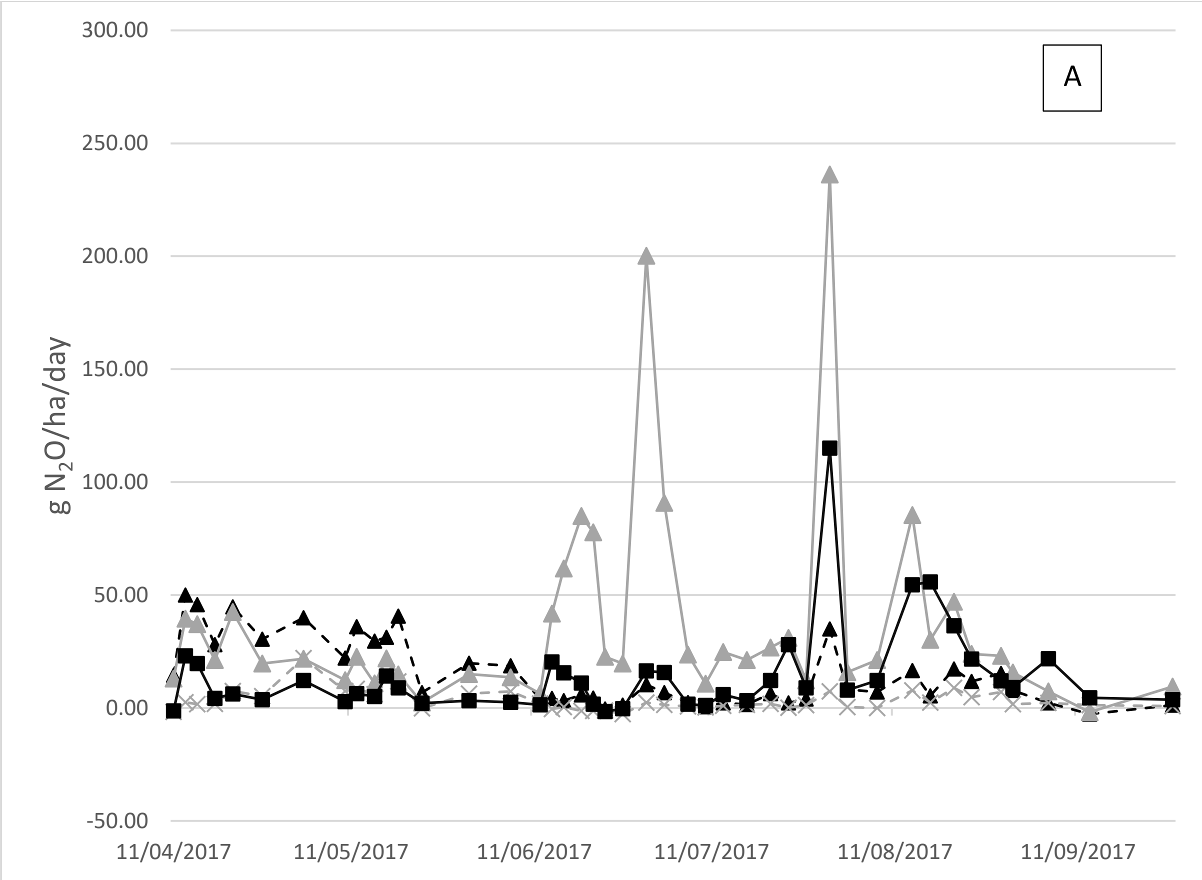
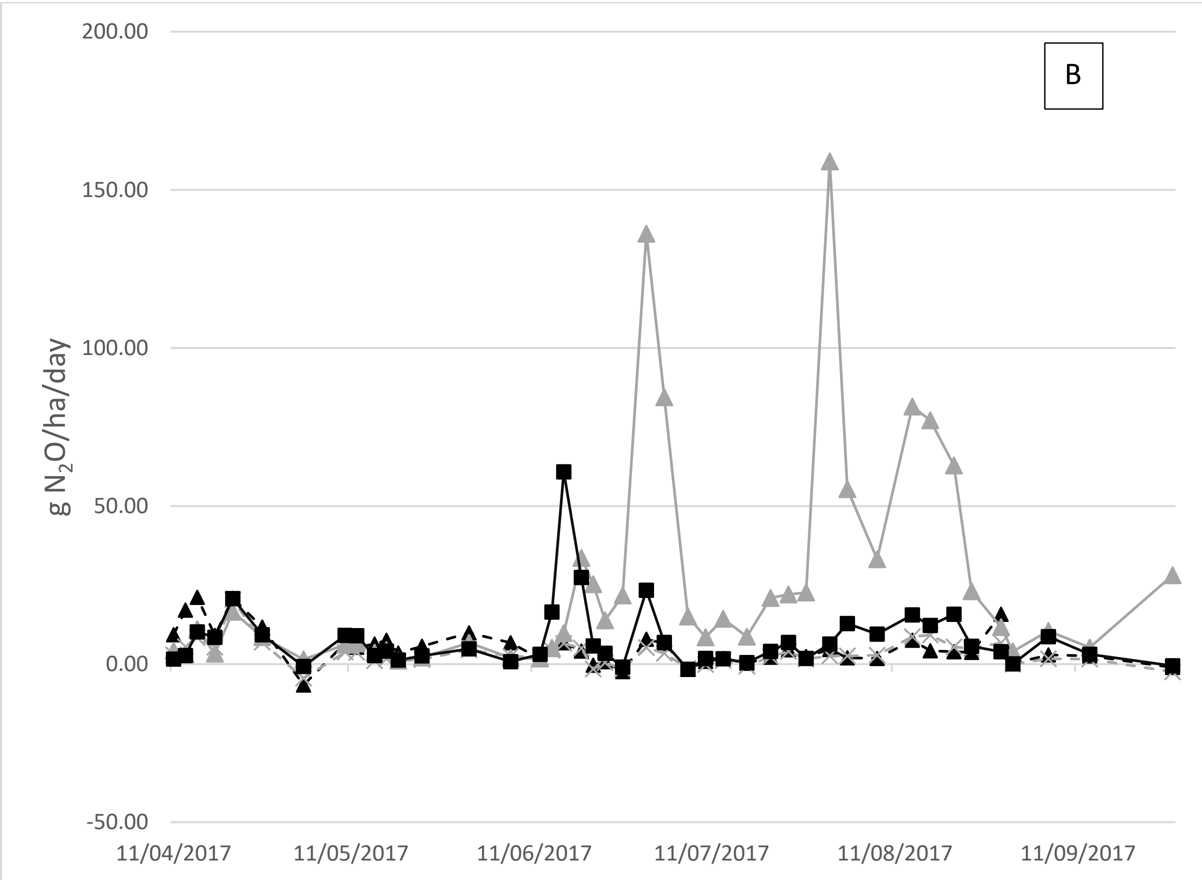


Figure 4.2. Mean daily temperature and rainfall during the sampling campaign observed at the meteorological station located on the NWFP. Treatment application occurred on 13/06/2017.


A
g N$_2$O/ha/day
300.00
250.00
200.00
150.00
100.00
50.00
0.00
-50.00
11/04/2017
11/05/2017
11/06/2017
11/07/2017
11/08/2017
11/09/2017


B
g N$_2$O/ha/day
200.00
150.00
100.00
50.00
0.00
-50.00
11/04/2017
11/05/2017
11/06/2017
11/07/2017
11/08/2017
11/09/2017

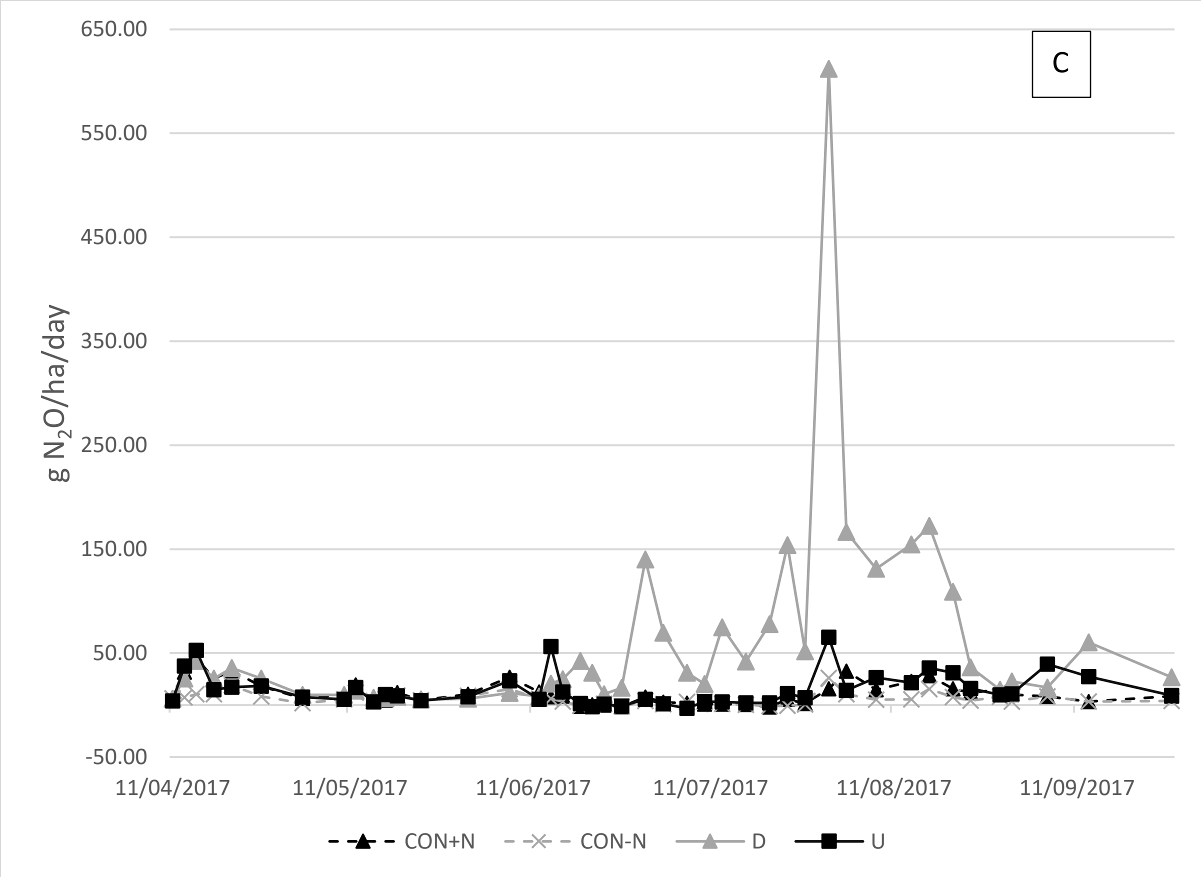


Figure 4.3. Daily N_2O fluxes for each farmlet. A: permanent pasture (PP); B: white clover/high sugar grass mix (WC); C: High sugar grass monoculture (HS). Note different y-axis scales. CON+N: control with N; CON-N: control without N; D: dung; U: urine.

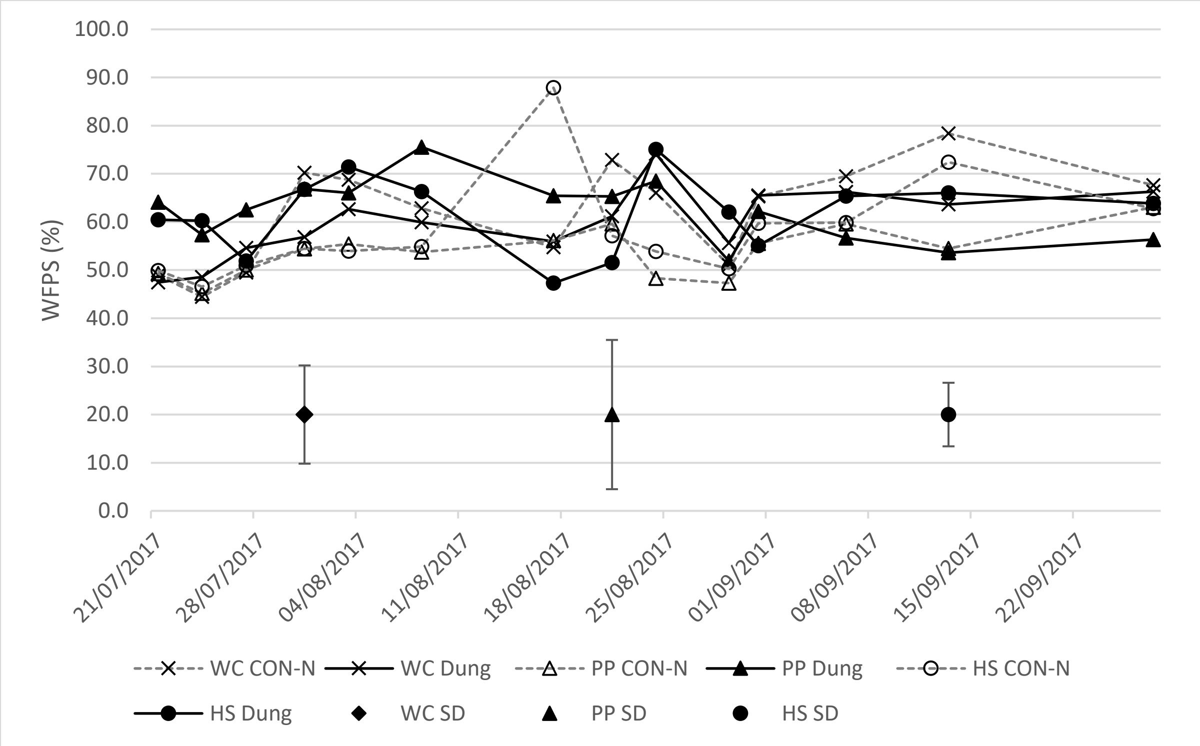


Figure 4.4. Water filled pore space (WFPS) from control without N (CON-N) and dung (D) plots post-treatment. Error bars represent the average standard deviation (SD) across replicates including both CON-N and D plots.

Chapter 5 – Impacts of adopting site-specific emission factors on uncertainties associated with life cycle assessment

5.1 Introduction

As discussed in Chapter 1, the method of life cycle assessment (LCA) has evolved and subsequently become a primary means to quantify environmental footprints of commodity value chains over the last few decades, and there is now a consensus that LCA studies must acknowledge uncertainties inherent within production systems to ensure their scientific robustness (Igos et al., 2018). Nonetheless, as discussed in Section 1.1, LCA studies often omit rigorous evaluation of system-level uncertainties (Imbeault-Tétreault et al., 2013), resorting instead to discrete scenario and sensitivity analyses or, more problematically, solely to point estimates (Curran, 2012). This is particularly problematic for analysis of livestock production systems, which are known to bear a high degree of uncertainties associated with their physical, chemical and biological processes that underpin meat, milk, egg and fibre production.

One of the most limiting aspects of compiling a life cycle inventory analysis (LCI) is uncertainty associated with emission factors (EF), or parameters linking nutrient inputs to GHG outputs from the system (Pouliot et al., 2012). On real-world livestock farms, many factors can affect these ratios, including weather, soil, plant/animal genetics, management practice (e.g. diet and housing) and interactions between them. The majority of carbon footprint studies, however, adopt EFs derived outside the actual system boundary, most commonly in the form of parameters defined as part of IPCC (2006) guidelines. As these "generic" EFs are designed to be applicable to a wide spectrum of production environments within an agroecological zone, a considerable level of uncertainty surrounds each of these values, as described in detail in Section 4.1.

The objective of the present chapter is to evaluate the impacts of adopting site-specific EFs on uncertainties associated with carbon footprints of livestock production systems. To achieve this, the two parameters for nitrous oxide (N_2O) emissions derived in Chapter 4, EF_1 (% fertiliser N lost as N_2O) and EF_3 (% urine and dung N deposited on pasture lost as N_2O), and an additional EF commonly known as Y_m, which characterises methane (CH_4) emissions through enteric fermentation (% gross energy intake emitted as CH_4), were integrated into an LCA model of the three NWFP enterprises for inter-system comparison. The resultant CFs and their confidence intervals were then compared against those obtained under the IPCC recommended EFs with a larger degree of uncertainty. Chapter 3 found that soil-originated N_2O and CH_4 arising from enteric fermentation, the major GHG emissions represented by these three parameters, collectively accounted for 48 – 51% of the system-wide CF when evaluated at the point estimates.

The NWFP (Section 1.4) was selected as the study site since its three-farmlet research design provides an ideal setting to investigate the research question discussed above. Compared to common grasses, white clover has been found to reduce enteric fermentation by grazing ruminants (Enriquez-Hidalgo et al., 2014, Hammond et al., 2011), although the level of this effect is highly heterogenous across production environments (van Dorland et al., 2007). On the other hand, high sugar grasses, or grass cultivars with elevated levels of water soluble carbohydrate (WSC), have been found to improve nitrogen use efficiency at the animal level (i.e. milk N per unit of dietary N) and are therefore believed to also improve system-wide nitrogen use efficiency (Miller et al., 2002), and reduce both N_2O emissions and nitrate (NO_3^-) losses per unit of final product (Soteriades et al., 2018). To make the matter more complex, these cultivars have contradicting results in relation to CH_4 emissions, with their relative performance vis-à-vis common grasses depending on sward composition, dry matter intake (DMI) and units chosen to express CH_4 (e.g. % gross energy or g CH_4/kg milk) (Ellis et al., 2012). However, as the majority of these results were obtained from zero-grazed studies to facilitate individual intake measurements, interactions between production environment, pasture/animal genetics and management are not well understood. EFs derived from farm-scale trials can overcome this limitation; to the best of my knowledge, this is the first LCA study of livestock production systems to utilise such values.

5.2 Materials and methods

5.2.1 Goal and scope definition

As discussed, the primary goal of the present study was to examine system-wide impacts of adopting site-specific emission factors on carbon footprints of pasture-based cattle production systems. In addition, an empirically motivated secondary goal was also defined to evaluate relative climate change impacts of three sward management strategies commonly observed in temperate grasslands; namely permanent pasture (PP), grass and white clover swards (WC) and monoculture grass leys using recommended grass varieties e.g. high-sugar grasses (HS) as discussed in detail in Section 1.4 and Chapter 3.

The study was carried out on the three cattle finishing enterprises (farmlets) at the NWFP and the adjacent breeding (cow/calf) farm that supplies weaned calves to them each year. Expanding the model that solely focused on the finishing farmlets developed in Chapter 3, the current system boundary covered both operations and was set as "cradle-to-gate" — from the production of raw materials to the departure of animals for slaughter **(Figure 5.1)**. The functional unit was set as 1 kg liveweight (LW, as opposed to 1 kg liveweight gain adopted in Chapter 3) and the downstream slaughtering process was excluded from the model. This decision was motivated by two factors: (1) to maximise cross-study comparability (McAuliffe et al., 2016); and (2) to minimise *off-farm* uncertainties in the model that may affect evaluation of *on-farm* uncertainties. All analyses were repeated for two generations of finishing cattle, those from grazing seasons 2015 (born 2014, *n* = 90) and 2016 (born 2015, *n* = 90).

5.2.2 Inventory analysis and impact assessment

5.2.2.1 Overall design

LCI and life cycle impact assessment (LCIA) were carried out according to ISO (2006) and BSI (2011) guidelines. CFs for each individual animal were calculated using the "animal-by-animal" framework devised in Chapter 3; however, the partial carbon footprint approach has now been extended to cradle-to-gate as described in Section 5.2.1. Inventory analysis for finishing enterprises (farmlets) utilised the NWFP's high-resolution records between 2014 (when the first calf was weaned) and 2017 (when the last calf was finished), as per the procedure outlined in Chapter 3. Amongst key variables, animal LW **(Table 5.1)** and pasture/silage quality **(Table 5.2)**, the latter determined by digestible energy (DE) and CP, were both measured every two to four weeks to estimate on-farm emissions during the corresponding period. For the breeding enterprise, a steady-state herd structure to supply 30 weaned calves was mathematically derived from the average parity number and the mortality rate measured on the farm **(Table A5.1)**, to account for the whole-herd emissions inclusive of pre-service heifers, those that replace the breeding stock, and those that fail to produce healthy calves. Since grazing fields on the breeding farm are permanent pasture and managed in a similar manner to the NWFP's permanent pasture (PP) farmlet, inputs to, and burdens arising from, a hectare of grassland (excluding animal-originated emissions) were duplicated from the latter's inventory. The finishing and breeding operations were subsequently linked to form a single LCA model **(Figure 5.1)**, with the final output (kg CO_2-eq/kg LW) representing emissions from both. As part of this process, ~28% of burdens arising from the breeding farm were allocated to sales of culled cows; given the considerably lower value of meat produced from these animals, the economic allocation method was adopted to split burdens between the two products. All results reported below are for production of prime suckler-beef only.

Similarly to Chapter 3, on-farm GHG emissions from cattle and pastures were calculated using a modified IPCC (2006) Tier 2 approach. To address the primary goal of the study, as outlined in Section 5.2.1, site-specific parameter values were obtained for EF_1, EF_3 **(Table 4.4)** and Y_m, the latter of which will be described in the next section. The values of EF_1 and EF_3 were shared between the PP farmlet and the breeding farm, which had comparable sward structures as discussed above. For all other EFs, including those for N_2O and CH_4 emissions from manure management and indirect N_2O emissions arising from leaching and atmospheric deposition, UK-specific values (Brown et al., 2018) were adopted universally across all farmlets and the breeding farm. On-farm impacts of sheep grazing, both positive through lamb production (and less pronouncedly through manure deposition) as well as negative through GHG emissions, were separated from the model using the decomposition method outlined in Chapter 3.

Emissions associated with background processes, such as the production and transport of straw for bedding and small quantities of supplementary feeds —soybean (*Glycine max*) meal in 2015 and rapeseed (*Brassica napus*) expeller meal in 2016 — were sourced from *Agri-footprint* V3 (Durlinger et al., 2017) and *ecoinvent* V3 (Wernet et al., 2016) databases. Inputs which accounted for <1% of system-wide emissions according to the analysis undertaken in Chapter 3 were assumed to be the same across both grazing seasons. CFs were calculated according to the IPCC (2013) 100-year average impact assessment method on SimaPro V8.2.3 (www.simapro.com).

5.2.2.2 Methane emission factors

In addition to EF_1 and EF_3 representing soil-originated N_2O emissions (derived in Chapter 4), Y_m representing animal-originated CH_4 emissions were also quantified on site. As CH_4 data were routinely recorded at the NWFP, their collection did not constitute part of the present PhD research; however, the author collated the raw data, then analysed and subsequently converted them to EFs that are suitable for LCA. Throughout the 2016/17 winter housing period (26/11/2016 – 12/04/2017, 137 days), all cattle on the NWFP's three farmlets were given access to the GreenFeed Emission Monitoring system (C-Lock Inc., Rapid City, SD), which dispenses a small quantity of concentrates to attract animals while measuring local CH_4 fluxes, at a maximum of three visits per day. Standard feed (silage produced from forages on the respective farmlet) and drinking water were given *ad libitum*. The measured level of gas concentration was first converted to daily CH_4 emissions for each animal on each day separately using the volumetric airflow rate and fractional air capture rate (Huhtanen et al., 2015). These values were then pooled across animals and the study period and, together with the gross energy intake estimated as part of the modified IPCC Tier 2 approach (Section 5.2.2.1), used to compute Y_m for each farmlet. The final values were 7.85%, 8.28% and 7.88% for PP, HS and WC, respectively.

5.2.3 Interpretation

5.2.3.1 Statistical analysis

As discussed in Chapter 3, one of the advantages of adopting an animal-by-animal framework for carbon footprints is that the treatment effect on CFs can be statistically tested using standard parametric procedures. Here again, statistical analyses were carried out in GenStat V18.1 (www.vsni.co.uk/software/genstat). Two-sample t-tests were used to test differences in CF between farmlets and between seasons. Correlations between CFs and their potential determinants were investigated through Pearson's correlation coefficient. The system-wide CF for each farmlet was quantified as the arithmetic mean across all animals.

5.2.3.2 Sensitivity analysis for model assumptions

As the NWFP is a farm-scale trial, its three farmlets are operated as commercial enterprises, with their management closely resembling private farms in neighbouring regions (Takahashi et al., 2018). The adjacent breeding farm, however, was established primarily to supply healthy calves to the NWFP and, therefore, subscribes to a more conservative strategy in terms of productivity than typically observed on more commercially driven operations. In particular, its stocking rate of ~1.3 livestock units (LU) per ha **(Table A5.1)** is at the lower end of the recommended range in England (AHDB, 2016), and the average parity number of ~4 births is also below common practice in temperate grasslands (Beauchemin et al., 2010). To examine the potential impacts of these "non-managerial" parameters on total CFs, two sets of scenario analyses were conducted. First, the pasture area was modified to move the stocking rate to the upper limit (2.5 LU/ha) and the lower limit (1 LU/ha) of AHDB guidelines. Second, the average parity number was altered to higher (6 calves/cow) and lower (2 calves/cow) levels from the baseline operation. For the latter analysis, the steady-state herd structure was recalculated under respective scenarios so that an equal number (30) of calves would be weaned and transferred to each NWFP farmlet regardless of the parameter choice. Finally, in line with ISO 14040 guidelines (ISO, 2006) that recommend a sensitivity analysis for the choice of allocation method, the baseline model was rerun under mass allocation to split burdens between culled cows and weaned calves.

5.2.3.3 Sensitivity and uncertainty analysis for site-specific emission factors

To meet the study's primary goal of examining the impacts of using farm-level EFs within the CF framework, a range of sensitivity and uncertainty analyses were carried out. First, site-specific EFs were replaced with default IPCC values for EF_1 (0.01), EF_3 (0.02) and Y_m (0.065). Second, to test the effect of local-level uncertainty surrounding measured EFs, high and low boundaries were placed independently around each of them. Following IPCC (2006) guidelines, these boundaries were set at −67% / +300% (EF_1 and EF_3), and ± 1 percentage point (Y_m); it is recognised, however, that an unidentified proportion of IPCC's confidence intervals is attributable to uncertainty regarding the location (e.g. weather and soil), and therefore the CF range produced from this analysis represents the widest possible values.

Furthermore, three sets of Monte Carlo (MC) analyses were also conducted to compare 95% confidence intervals of CFs derived with and without the existence of site-specific EFs. In the first analysis (MC1), the measured values of EF_1, EF_3 and Y_m were assumed to be *uncertain*, and given the standard probability distributions (IPCC, 2006) around the measured values. In the second analysis

(MC2), the same parameters were assumed to be *completely certain*, and excluded from random drawings at each iteration. As discussed, MC1 is likely to overestimate the degree of local-level uncertainty (when the location of the farm is known), with the true range expected to lie between the distributions derived by these two methods. Finally, confidence intervals that would be derived in the absence of site specific EFs were also estimated (MC3), using default IPCC values (listed above) and probability distributions. All sensitivity and uncertainty analyses were carried out using 2016 data and, following the method developed in Chapter 3, performed on the 1^{st} (best), 15^{th} (median) and 30^{th} (worst) animals from each farmlet based on their GWP rankings — to detect potential interactions between model parameters and animal genetics. All MC simulations were conducted using SimaPro V8.2.3, with 1000 iterations performed per animal for each scenario.

5.3 Results and discussion

5.3.1 Baseline results

Across the three treatments, the white clover mixed sward (WC) had significantly lower CF than PP and the high sugar grass monoculture (HS) in both 2015 and 2016 ($p < 0.001$ in all four pairwise cases, **Table 5.4**). This finding is consistent with what was reported in Chapter 3 and primarily driven by little (2015) or no (2016) use of synthetic N fertiliser, of which production is estimated to generate 8.8 kg CO_2-eq/kg N. In the current study, however, this saving was more muted due to the inclusion of a permanent pasture-based breeding farm. While it is reasonable to conjecture that switching the latter to a white clover mixed sward will further reduce the overall CFs, the risk of failing to establish a vigorous and spatially homogenous sward must be carefully considered before universal recommendations are made. System-wide, GWP attributable to the suckler enterprise accounted for 61-67% of the total value, of which enteric CH_4 from breeding cows and burdens arising from ammonium nitrate (NH_4NO_3) production were the largest contributors **(Tables A5.2 and A5.3)**. The high burden associated with the breeding stage led the 2016 herd to record generally higher levels of CFs than the 2015 herd ($p = 0.037$), as the former spent more time with their mothers and were weaned heavier **(Table 5.1)**; as a result, the allocation rate *vis-à-vis* culled cows became higher as well.

Intra-system differences in environmental performance were largely explained by livestock performance, again supporting an earlier assertion made in Chapter 3. The overall CFs were weakly but negatively correlated with lifetime average daily gains **(Figure A5.1)**, with correlation coefficients of −0.21 ($p = 0.11$), −0.33 ($p = 0.010$) and −0.14 ($p = 0.28$) for PP, WC and HS, respectively. More importantly, average daily gains post-weaning showed strong and negative correlations with GWP attributable to the finishing process **(Figure A5.2)**, with correlation coefficients of −0.81 ($p < 0.001$), −0.74 ($p < 0.001$) and −0.59 ($p < 0.001$) respectively. It was also reiterated that sex is a strong determinant of GWP, with steers recording significantly ($p < 0.001$) lower CFs than heifers for all farmlets and years **(Figure A5.3)**.

5.3.2 Sensitivity analysis for model assumptions

Results of the sensitivity analyses suggest that changing the attitude towards the risk facing the breeding enterprise has some environmental implications. When its stocking rate was raised to the upper limit of current industrial guidelines in England (AHDB, 2016), CF decreased by approximately 13% across the three farmlets **(Figures 5.2, A5.4 and A5.5)**. Reducing the stocking rate to the lower limit of 1 LU/ha, on the other hand, increased emissions by 7%. Increasing the average parity number by two births (to six calves) reduced CF by up to 21%, while decreasing by two births increased emissions by > 100% in most cases. Although the latter scenario is unlikely to be observed in the real world even amongst the most conservative producers, the highly asymmetric (nonlinear) nature of these impacts is important to note; unlike at the NWFP, low parity numbers on commercial farms are often a result of health issues (e.g. mastitis) rather than the producer's risk aversion, and above results indicate that their environmental burdens become exponentially high as replacement rates rise. When the high stocking rate was combined with the increased parity, GWP decreased by 29%, resulting in 17.9, 16.0 and 18.2 kg CO2-eq/kg LW for median animals on PP, WC and HS farmlets, respectively. For the reasons outlined in Section 5.2.3.2, these values would likely represent commercial pasture-based beef farms in Southwest England that subscribe to best management practices. It should be noted, however, that GWP estimates derived under the animal-by-animal framework are higher than those derived under a representative animal approach, due to the latter's tendency to systematically underestimate emissions associated with poorly performing animals (as demonstrated in Chapter 3). Finally, when environmental burdens arising from the breeding farm were allocated between culled cows and weaned calves based on mass rather than economic value, total GWP reduced by 7%, suggesting that some findings may be moderately sensitive to the choice of allocation method. Nonetheless, given the large quality difference between the two products, interpretation of mass-allocated results is not straightforward.

5.3.3 Sensitivity and uncertainty analysis for site-specific emission factors

Replacing the site-specific parameters for N_2O emissions (EF_1 and EF_3) with default IPCC (2006) values did not have any notable impacts on overall GWP, with changes accounting for a 1% reduction in total emissions **(Figure 2 and Table A5.4)**. Applying the IPCC uncertainty range over site-specific EF_1 and EF_3, however, resulted in reversals of rankings amongst farmlets. For instance, when the upper limit value (+300%) was assumed under the median animal, the HS system, which recorded the highest CF in the baseline analysis, performed slightly better than PP system **(Table A5.4)**. This change was driven by the lower EF_1 value for HS, which more than offset the higher EF_3 value on HS under the extreme scenario **(Figure 5.2)**. In other words, the adoption of upper limit values magnified the quantitative impact of inter-system differences in EF_1 on resultant GWP values; generic EFs, which do not discriminate between different grasses and legumes, are unable to detect these changes.

Contrary to the case with N_2O EFs, a considerable GWP effect was observed when measured Y_m values were replaced by default values. Total emissions decreased by around 8%, demonstrating that the discrepancy between local and generic EFs was large enough to translate into system-wide environmental performance. More critically, site-specific Y_m values derived in the current study (0.079 – 0.083) indicated the possibility that the IPCC uncertainty boundary (0.055 – 0.075) may not be wide enough to capture a complete spectrum of common farming systems, something acknowledged for tropical regions but not for temperate regions by IPCC (2006). As noted, current Y_m values are based on single-year data, further research is required before a firm conclusion can be drawn on this matter. Amongst the three farmlets, derived Y_m values were highest under HS, corroborating the earlier report by Ellis et al. (2012). Consequently, the relative environmental performance of this system (as evaluated by the CF difference against PP and WC) "improved" when the generic EF was adopted, for the same reason as that described above. Under site-specific EFs, HS system performed worse, again relatively compared to other systems, under less efficient animals that required more gross energy over their lifetime. The last finding demonstrates that there are interactions between animal genetics and sward sturucture, and therefore decisions on animal selection, pasture type and management strategies should be made at the same time.

The results of the three MC experiments showed that measurements of site-specific EFs have a large potential to reduce the overall uncertainty of carbon footprints. Across all farmlets and all animals (best, median and worst), MC1 produced a 95% confidence interval of −11% to 17% around the baseline GWP (24.4 kg CO_2-eq/kg LW). This range was narrowed to −6% to 10% under MC2, whereby EF_1, EF_3 and Y_m were assumed to be completely certain. The reduction of uncertainty associated with complete knowledge of the three parameters was 43%, or a range equivalent to 3.0 kg CO_2-eq/kg LW. When site-specific information was removed altogether under MC3, not only did the confidence interval become wider (5.3 CO_2-eq/kg LW) but also skewed to the left. The latter result suggests that the effect of site-specific Y_m (all larger than generic values) was stronger than that of EF_1 and EF_3 (generally smaller than generic values), in part because of larger burdens associated with enteric fermentation, and in part because of the larger discrepancy between local and generic values.

5.3.4 Comparisons with previous beef studies

Although care should be taken when comparing results from multiple LCA studies with different system boundaries (McAuliffe et al., 2016) and model assumptions (Roy et al., 2009), it is worthwhile contextualising the results of the present study within the current understanding of climate change impacts brought about by the beef sector. The baseline CFs from the NWFP's three systems are notably higher than previous research adopting the same functional unit of 1 kg LW **(Table 5.4)**. For example, an Irish study by Casey and Holden (2006) reports a GWP of 11.3 kg CO2-eq/kg LW, approximately half the value reported here under a "business as usual" scenario largely comparable to English pasture-based systems (but with slightly a higher level of concentrates-use). The same statement is true for a Danish and Swedish study by Mogensen et al. (2015), which uses 1 kg of carcase weight as the functional unit, when the present results are converted using the locally measured kill-out percentage of 51%. Although part of these differences can be explained by the necessity to employ a conservative strategy at the breeding enterprise (Section 5.2.3.2), CF estimates after correcting for this requirement were still ~35% higher than Casey and Holden (2006). These differences, therefore, would be attributable to a combination of site-specific EFs, downward bias inherent in models built with national and regional statistics (Section 5.3.2), geographical differences with regards to pasture and animal productivity, optimality of management and, finally, differences in model assumption. Of the latter point, rigorous use of herd dynamics models (Section 5.2.2.1), as opposed to a snapshot herd population, seems relatively rare in the literature, and this could be causing another bias amongst some studies; however, evidence is insufficient to firmly reach this conclusion, as detailed assumptions behind the herd structure are often unreported in LCA studies.

5.4 Conclusion

This chapter used locally measured EFs to calculate the full CF for three pasture-based beef production systems currently trialled on the NWFP and compare the derived results to what would have been obtained without site-specific information. The results suggested that EFs specific to the location and management strategies have strong potential to reduce uncertainty associated with carbon footprints. In particular, the proposed approach successfully accounted for the environmental impacts caused by interactions between animal genetics and sward sturucture, which would have been undetected under a modelling framework employing generic EFs. Based on this finding, it was suggested that farming strategies encompassing animal selection and pasture management should be jointly designed, so as to maximise the synergy between both genetic resources. Future studies should further investigate methods to decompose uncertainty about the location of the farm from uncertainty that exists within that specific location, with the view to identify the cost-benefit relationship for collecting site-specific information.

While discussions so far in the literature, and therefore this thesis, have primarily been focusing on uncertainties associated with measures of environmental burdens, or the numerators of LCA outputs, their denominators — the functional units — also carry a considerable level of uncertainties. Shifting focus from *quantity*-based LCA to *quality*-based LCA, the next and final chapter considers how nutritional differences in our food should play a role when determining the overall environmental footprints of animal-based products.

Table 5.1. Livestock performance.

		WC[a]		PP[b]		HS[c]	
	Unit	2015	2016	2015	2016	2015	2016
Weaning weight	kg	279	330	279	332	284	333
Finishing weight	kg	582	624	607	625	590	633
Total growth	kg	537	578	562	578	545	585
Time spent in SH[d]	d	193	224	184	217	203	220
Time spent in FH[e]	d	448	395	435	408	439	401
Age at slaughter	d	642	618	619	626	642	620
Average daily gain (SH[d])	kg/d	1.24	1.27	1.34	1.31	1.21	1.30
Average daily gain (FH[e])	kg/d	0.68	0.72	0.76	0.69	0.70	0.73
Average daily gain (total)	kg/d	0.84	0.94	0.91	0.93	0.85	0.95

[a] WC: White clover and high sugar grass mix sward
[b] PP: Permanent pasture
[c] HS: High sugar grass monoculture
[d] SH: Suckler-herd
[e] FH: Finishing-herd

Table 5.2. Life cycle inventory for finishing systems (average across 2015 and 2016).

	Unit	WC[a]	PP[b]	HS[c]
Area	ha	20.85	21.61	21.45
Fertiliser area	ha	20.52	21.24	21.03
FYM area	ha	17.97	18.90	18.34
Yield	kg DM/ha	10470	11671	10761
Fertiliser				
N	kg	341	4153	3819
P	kg	762	232	289
K	kg	666	876	569
Lime	kg	2551	1916	5941
Rapeseed expeller meal[d]	kg	3875	1927	5697
Soybean[e]	kg	651	651	672
Straw	kg	39589	38824	39484
Transport				
Straw (road)	tkm	2478	2430	2472
Fertiliser (road)	tkm	1849	3071	4544
Rapeseed expeller meal[d] (road)	kg	3875	1927	5697
Soybean (sea)[e]	tkm	6267	6267	6469
Soybean (road)[e]	tkm	155	155	160
Pasture quality				
DE	%	77.16	76.205	75.73
CP	%	21.45	21.91	18.98
Silage quality				
DE	%	66.50	67.58	66.05
CP	%	11.15	13.45	12.55

[a] White clover and high sugar grass mix sward
[b] Permanent pasture
[c] High sugar grass monoculture
[d] Only supplemented in 2016
[e] Only supplemented in 2015

Table 5.3. Summary breakdowns of system-wide GWP.

	2015			2016		
	WC[a]	PP[b]	HS[c]	WC[a]	PP[b]	HS[c]
Suckler herd						
Methane (enteric fermentation)	6.22	5.95	6.22	6.86	6.87	6.84
Methane (manure management)	0.92	0.88	0.92	1.02	1.01	1.00
Pasture emissions	5.47	5.23	5.47	6.02	6.04	6.01
Others	1.40	1.34	1.39	1.53	1.56	1.55
Finishing herd						
Methane (enteric fermentation)	4.84	4.61	4.97	4.16	4.29	4.61
Methane (manure management)	0.96	0.74	0.87	0.67	0.68	0.78
Pasture emissions	1.24	3.85	3.11	0.79	2.84	2.42
Others	1.84	1.62	1.86	1.94	1.75	2.17
Total	22.88	24.23	24.82	22.99	25.05	25.39

[a] White clover mixed sward

[b] Permanent pasture

[c] High sugar grass monoculture

Table 5.4. Comparison of results with previously published beef production research.

Study	Scope	Functional unit	GWP (CO_2-eq)
Casey and Holden (2006) (IRE)	Production of raw materials to departure from the farm gate	1 kg LW[a]	11.3 kg
Pelletier et al. (2010) (USA)	Production of raw materials to departure from the farm gate	1 kg LW[a]	14.8 kg
Nguyen et al. (2010a) (EU)	Production of raw materials to departure from the farm gate	1 kg CW[b]	27.3 kg
Beauchemin et al. (2010) (CAN)	Production of raw materials to departure from the farm gate	1 kg CW[b]	22.0 kg
Peters et al. (2010) (AUS)	Production of raw materials to departure from the slaughterhouse gate	1 kg HSCW[c]	12.0 kg
Roop et al. (2013) (USA)	Production of raw materials to departure from the farm gate	1 kg LW[a]	13.8 kg
Ridoutt et al. (2011) (AUS)	Production of raw materials to departure from the farm gate	1 kg LW[a]	10.2 kg
Wiedemann et al. (2015) (AUS)	Production of raw materials to departure from wholesale warehouse gate	1 kg beef product	23.4 kg
Mogensen et al. (2015) (DEN/SWE)	Production of raw materials to departure from the farm gate	1 kg CW[b]	23.1 kg
Dick et al. (2015) (BRA)	Production of raw materials to departure from the farm gate	1 kg LWG[a]	22.5 kg
Current study (UK) – PP, baseline	Production of raw materials to departure from the farm gate	1 kg LW[a]	24.6[d] kg
Current study (UK) – WC, baseline			22.9[d] kg
Current study (UK) – HS, baseline			25.1[d] kg
Current study (UK) – PP, improved efficiency			17.9[d] kg
Current study (UK) – WC, improved efficiency			16.0[d] kg
Current study (UK) – HS, improved efficiency			18.2[d] kg

[a] Liveweight (gain); [b] Carcase weight; [c] Hot standard carcase weight.
[d] All values are averaged across 2015 and 2016.

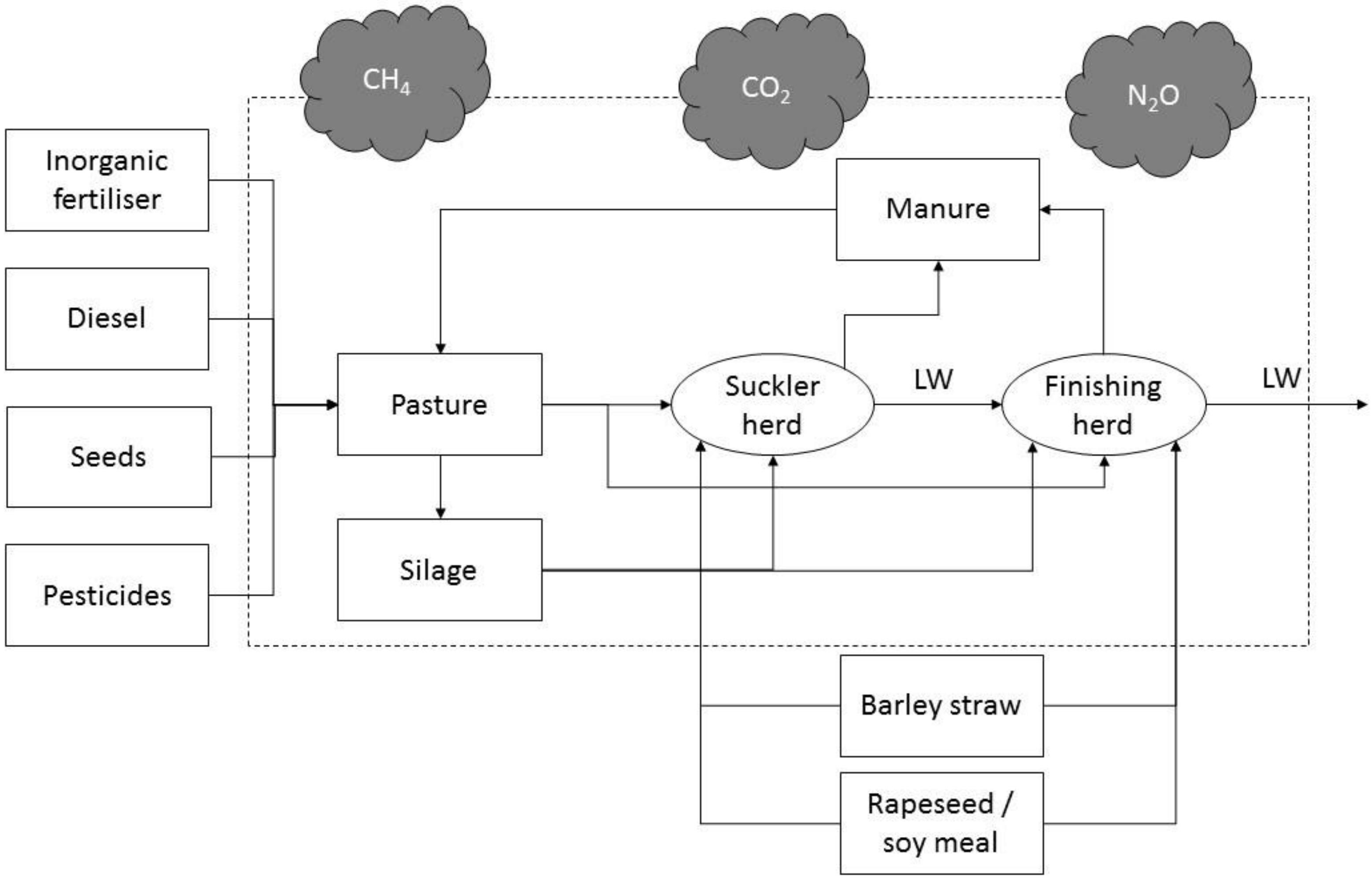


Figure 5.1. System boundary of the beef system. For emissions attributable to foreground processes (inside the dotted line), primary data were collected from the farm.

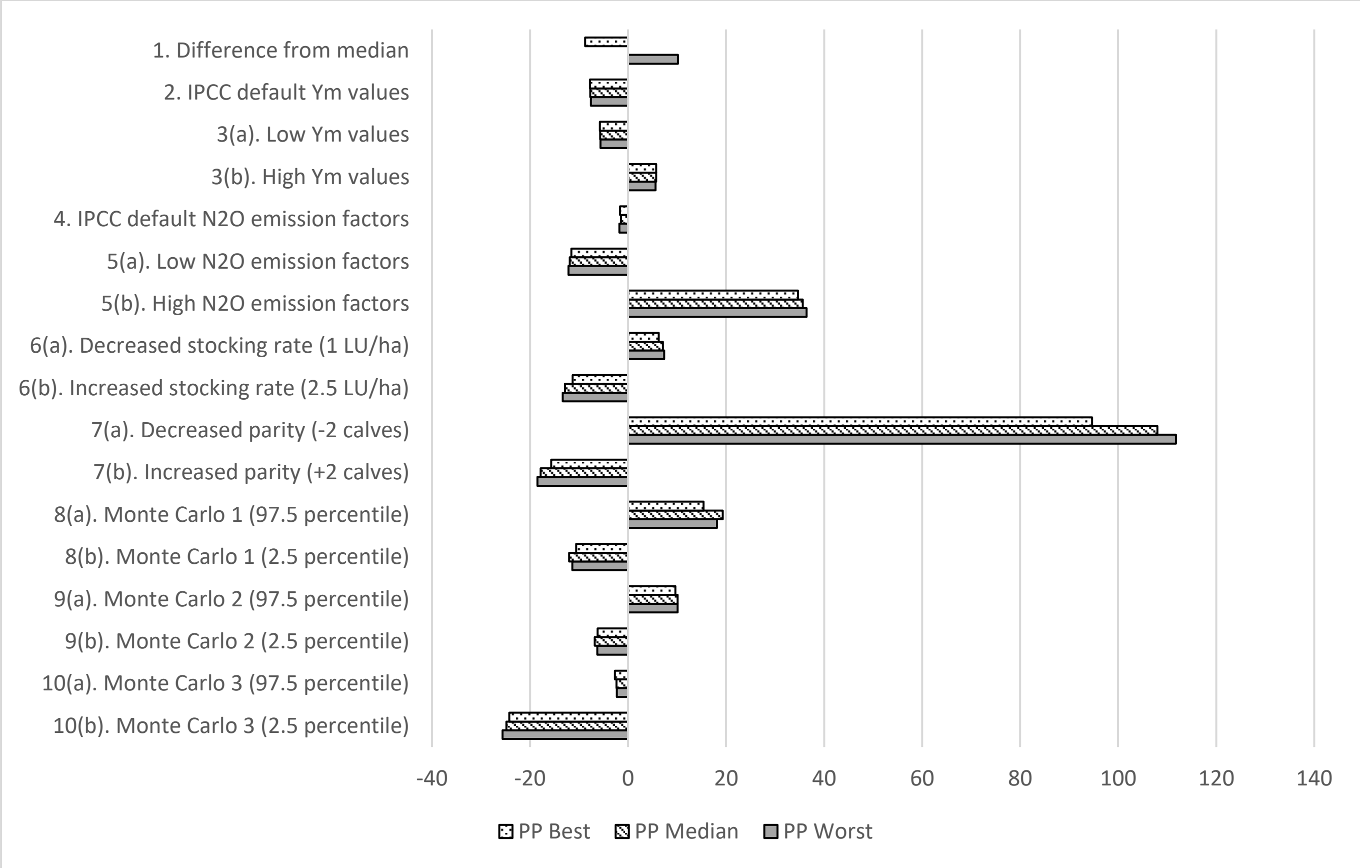


Figure 5.2. Results of scenario and sensitivity analyses carried out for the best (1st), median (15th) and worst (30th) animals on the permanent pasture (PP) system based on their GWP rankings. All values are expressed as percentage differences from the baseline results for the same animals.

Appendix to Chapter 5.

Table A5.1. Breeding herd structure and performance indicators.

Parameter	Unit	Value
Cows	n	127.8
Heifers	n	71.5
Calves	n	90.0
Pasture area	ha	137.1
Replacement rate	%	23.6
Cow mortality	%	3.7
Cow bodyweight[a]	kg	675
Heifer bodyweight[a]	kg	488
Calf bodyweight[a]	kg	208
Lifetime parity (calves per cow)	n	4
Culled cow (total liveweight departing the herd for slaughterhouse)	kg	17,342
Weaned cattle (total liveweight departing the herd for finishing enterprise)	kg	29,880

[a] Average bodyweight of an animal during the time defined as “cow”, “heifer” and “calf”.

Table A5.2. Detailed breakdown of contributors to global warming potential in the 2015 production cycle (kg CO_2-eq/kg liveweight).

Parameter	PP	(Range)	HS	(Range)	WC	(Range)
Suckler herd						
Methane (enteric fermentation)						
Breeding heifers	0.67	(0.56 - 0.80)	0.70	(0.57 - 0.89)	0.70	(0.56 - 0.86)
Calves	0.83	(0.70 - 0.99)	0.87	(0.71 - 1.10)	0.87	(0.69 - 1.07)
Cows	4.45	(3.71 - 5.29)	4.65	(3.76 - 5.87)	4.65	(3.68 - 5.72)
Methane (manure management)						
Breeding heifers	0.10	(0.08 - 0.12)	0.10	(0.08 - 0.13)	0.10	(0.08 - 0.13)
Calves	0.12	(0.10 - 0.15)	0.13	(0.11 - 0.16)	0.13	(0.10 - 0.16)
Cows	0.66	(0.55 - 0.79)	0.69	(0.56 - 0.87)	0.69	(0.55 - 0.85)
Direct nitrous oxide (manure management)						
Breeding heifers	0.04	(0.04 - 0.05)	0.05	(0.04 - 0.06)	0.05	(0.04 - 0.06)
Calves	0.05	(0.05 - 0.07)	0.06	(0.05 - 0.07)	0.06	(0.05 - 0.07)
Cows	0.30	(0.25 - 0.36)	0.31	(0.25 - 0.40)	0.31	(0.25 - 0.39)
Straw production	0.28	(0.23 - 0.33)	0.29	(0.24 - 0.37)	0.29	(0.23 - 0.36)
Pasture emissions						
Ammonium nitrate production	2.63	(2.19 - 3.13)	2.75	(2.22 - 3.47)	2.75	(2.17 - 3.38)
Nitrous oxide from synthetic N fertiliser application	1.72	(1.43 - 2.04)	1.79	(1.45 - 2.27)	1.79	(1.42 - 2.21)
Nitrous oxide from manure	0.72	(0.60 - 0.86)	0.76	(0.61 - 0.96)	0.76	(0.60 - 0.93)
Nitrous oxide from leaching	0.16	(0.14 - 0.20)	0.17	(0.14 - 0.22)	0.17	(0.14 - 0.21)
Others[a]	0.66	(0.53 - 0.83)	0.68	(0.52 - 0.86)	0.69	(0.54 - 0.86)
Sub-total	13.41	(11.20 - 16.00)	14.00	(11.30 - 17.70)	14.01	(11.10 - 17.20)
Finishing herd						
Methane (enteric fermentation)	4.61	(3.93 - 5.12)	4.97	(3.48 - 5.67)	4.84	(4.05 - 5.54)
Methane (manure management)	0.74	(0.37 - 0.94)	0.87	(0.62 - 1.13)	0.96	(0.57 - 1.32)
Direct nitrous oxide (manure management)	0.62	(0.57 - 0.68)	0.56	(0.39 - 0.62)	0.55	(0.51 - 0.60)
Straw production	0.30	(0.25 - 0.34)	0.31	(0.28 - 0.36)	0.32	(0.27 - 0.37)
Pasture emissions						
Ammonium nitrate production	1.90	(1.60 - 2.14)	1.72	(1.51 - 1.95)	0.27	(0.15 - 0.31)
Nitrous oxide from synthetic N fertiliser application	1.24	(1.05 - 1.40)	0.59	(0.52 - 0.67)	0.16	(0.13 - 0.18)
Nitrous oxide from grazing urine and dung	0.45	(0.38 - 0.49)	0.64	(0.54 - 0.72)	0.58	(0.23 - 0.31)
Nitrous oxide from manure	0.26	(0.15 - 0.33)	0.17	(0.12 - 0.21)	0.23	(0.49 - 0.65)
Others[a]	0.70	(0.54 - 0.84)	0.99	(0.85 - 1.11)	0.97	(0.85 - 1.19)
Sub-total	10.82	(9.70 - 12.10)	10.82	(8.80 - 12.10)	8.87	(7.50 - 9.90)
Total	24.23	(22.20 - 26.50)	24.82	(22.60 - 28.00)	22.88	(20.50 - 26.40)

[a] Total value from processes which contributed < 1% to overall global warming potential.

Table A5.3. Detailed breakdown of contributors to total global warming potential in the 2016 production cycle (kg CO_2-eq/kg LW).

Parameter	PP	(Range)	HS	(Range)	WC	(Range)
Suckler herd						
Methane (enteric fermentation)						
Breeding heifers	0.78	(0.62 - 0.94)	0.77	(0.60 - 0.95)	0.77	(0.66 - 0.91)
Calves	0.96	(0.77 - 1.16)	0.96	(0.74 - 1.18)	0.96	(0.82 - 1.12)
Cows	5.14	(4.09 - 6.21)	5.11	(3.95 - 6.28)	5.12	(4.40 - 5.99)
Methane (manure management)						
Breeding heifers	0.12	(0.09 - 0.14)	0.11	(0.09 - 0.14)	0.11	(0.10 - 0.13)
Calves	0.14	(0.11 - 0.17)	0.14	(0.11 - 0.18)	0.14	(0.12 - 0.17)
Cows	0.75	(0.33 - 0.92)	0.75	(0.34 - 0.93)	0.76	(0.65 - 0.89)
Direct nitrous oxide (manure management)						
Breeding heifers	0.05	(0.04 - 0.06)	0.05	(0.04 - 0.06)	0.05	(0.04 - 0.06)
Calves	0.06	(0.05 - 0.08)	0.06	(0.05 - 0.08)	0.06	(0.05 - 0.07)
Cows	0.35	(0.28 - 0.42)	0.36	(0.27 - 0.93)	0.35	(0.30 - 0.41)
Straw production	0.32	(0.26 - 0.39)	0.32	(0.25 - 0.40)	0.32	(0.28 - 0.38)
Pasture emissions						
Ammonium nitrate production	3.03	(2.42 - 3.67)	3.02	(2.34 - 3.71)	3.03	(2.60 - 3.54)
Nitrous oxide from synthetic N fertiliser application	1.98	(1.58 - 2.40)	1.97	(1.53 - 2.43)	1.98	(1.70 - 2.31)
Nitrous oxide from manure	0.84	(0.67 - 1.01)	0.83	(0.64 - 1.02)	0.83	(0.72 - 0.98)
Nitrous oxide from leaching	0.19	(0.15 - 0.23)	0.19	(0.15 - 0.23)	0.19	(0.16 - 0.22)
Others[a]	0.77	(0.57 - 1.16)	0.75	(0.38 - 1.19)	0.75	(0.62 - 0.93)
Sub-total	15.48	(12.30 - 18.70)	15.41	(11.90 - 18.90)	15.43	(13.30 - 18.10)
Finishing herd						
Methane (enteric fermentation)	4.29	(3.38 - 5.13)	4.61	(3.63 - 5.36)	4.16	(3.33 - 4.96)
Methane (manure management)	0.68	(0.47 - 0.91)	0.78	(0.60 - 1.02)	0.67	(0.49 - 0.92)
Direct nitrous oxide (manure management)	0.68	(0.55 - 0.79)	0.59	(0.47 - 0.68)	0.60	(0.49 - 0.71)
Rapeseed meal	0.10	(0.25 - 0.35)	0.30	(0.23 - 0.33)	0.21	(0.24 - 0.34)
Straw production	0.29	(0.09 - 0.13)	0.29	(0.24 - 0.39)	0.29	(0.17 - 0.24)
Pasture emissions						
Ammonium nitrate production	1.25	(1.06 - 1.52)	1.22	(0.97 - 1.39)	NA	NA
Nitrous oxide from synthetic N fertiliser application	0.82	(0.69 - 0.99)	0.42	(0.34 - 0.48)	NA	NA
Nitrous oxide from grazing urine and dung	0.43	(0.37 - 0.50)	0.61	(0.54 - 0.70)	0.54	(0.46 - 0.60)
Nitrous oxide from manure	0.35	(0.25 - 0.45)	0.17	(0.14 - 0.22)	0.25	(0.19 - 0.34)
Others[a]	0.68	(0.57 - 0.81)	0.98	(0.76 - 1.12)	0.83	(0.71 - 0.95)
Sub-total	9.57	(8.00 - 10.90)	9.98	(8.40 - 11.40)	7.56	(6.30 - 8.80)
Total	25.05	(22.70 - 27.40)	25.39	(22.30 - 28.20)	22.99	(20.90 - 24.80)

[a] Total value from processes which contributed < 1% to overall global warming potential.

Table A5.4. Detailed results of sensitivity and scenario analyses (kg CO_2-eq/kg liveweight).

	PP[a]			WC[b]			HS[c]		
Animal efficiency	Median	Best	Worst	Median	Best	Worst	Median	Best	Worst
1. Baseline	24.9	22.7	27.4	23.1	20.9	24.8	25.2	22.3	28.2
2. IPCC default Y_m[d] values	23.0	20.9	25.3	21.1	19.2	22.8	23.0	20.4	26.0
3(a). Low Y_m[d] values	23.5	21.4	25.9	21.6	19.6	23.3	23.7	21.1	26.7
3(b). High Y_m[d] values	26.3	24.0	29.0	24.5	22.2	26.3	26.6	23.6	29.8
4.IPCC default N_2O emission factors	24.5	22.3	26.9	22.7	20.7	24.4	25.1	22.4	28.1
5(a). Low N_2O emission factors	21.9	20.1	24.1	20.5	18.7	22.0	22.4	20.0	25.0
5(b). High N_2O emission factors	33.7	30.6	37.4	30.6	27.6	33.2	33.5	29.3	38.1
6(a). Decreased stocking rate (1 LU[e]/ha)	26.7	24.1	29.4	24.8	22.5	26.8	26.9	23.7	30.4
6(b). Increased stocking rate (2.5 LU[e]/ha)	21.7	20.1	23.8	19.8	18.2	21.1	22.0	19.9	24.3
7(a). Decreased parity (-2 calves)	51.8	44.2	58.1	50.2	44.0	55.6	51.7	43.1	61.3
7(b). Increased parity (+2 calves)	20.4	19.2	22.4	18.6	17.1	19.7	20.8	18.9	22.8
8(b). MC[f] 1 (2.5 percentile)	21.9	20.3	24.3	20.6	18.8	22.4	22.6	20.0	24.9
8(a). MC[f] 1 (97.5 percentile)	29.7	26.2	32.4	26.9	24.3	29.5	29.2	25.5	33.8
9(b). MC[f] 2 (2.5 percentile)	23.2	21.3	25.7	21.8	19.1	23.7	23.6	20.9	26.4
9(a). MC[f] 2 (97.5 percentile)	27.4	24.9	30.2	24.9	22.7	27.4	27.6	24.5	31.1
10(b). MC[f] 3 (2.5 percentile)	18.7	17.2	20.4	17.2	15.7	18.2	19.0	17.2	21.2
10(a). MC[f] 3 (97.5 percentile)	24.3	22.1	26.8	21.9	19.9	23.8	24.7	21.8	27.6

[a] Permanent pasture
[b] White clover mixed sward
[c] High sugar grass monoculture
[d] Methane conversion factors
[e] Livestock unit
[f] Monte Carlo simulations

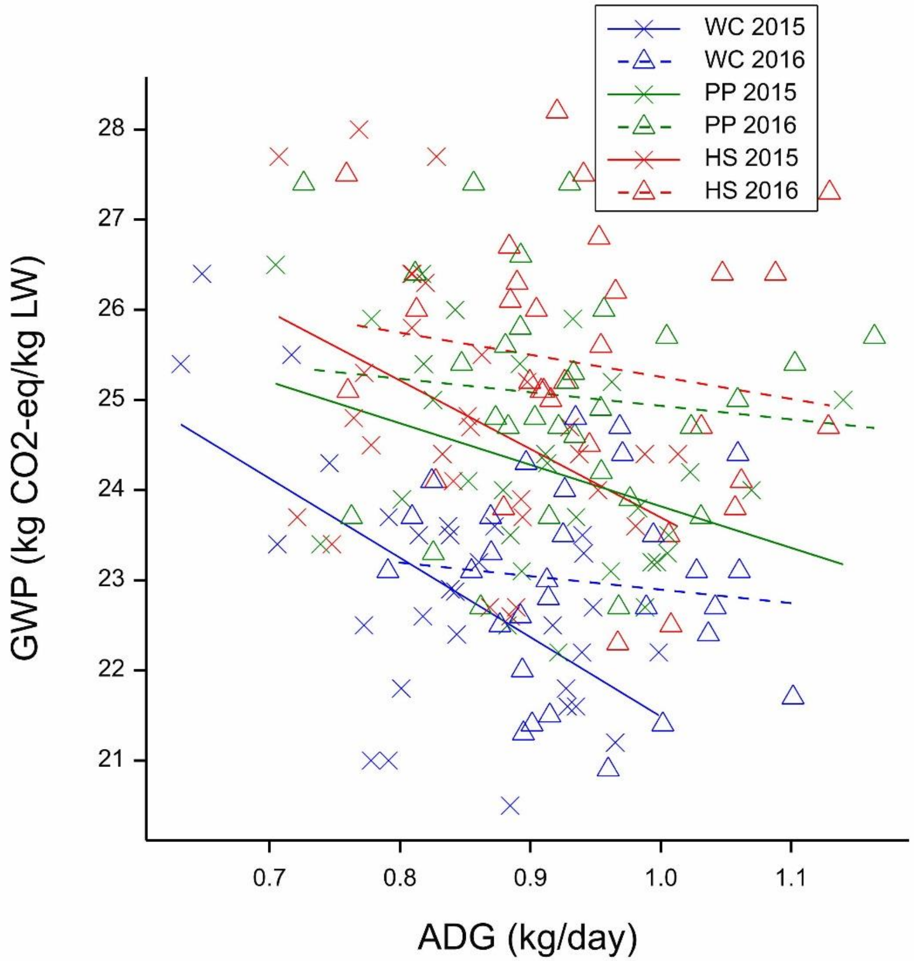


Figure A5.1. Relationship between individual animals' carbon footprints (as GWP per kg LW) and average daily gains (ADG) during their lifetime. PP: permanent pasture; WC: white clover mix; HS: high sugar grass.

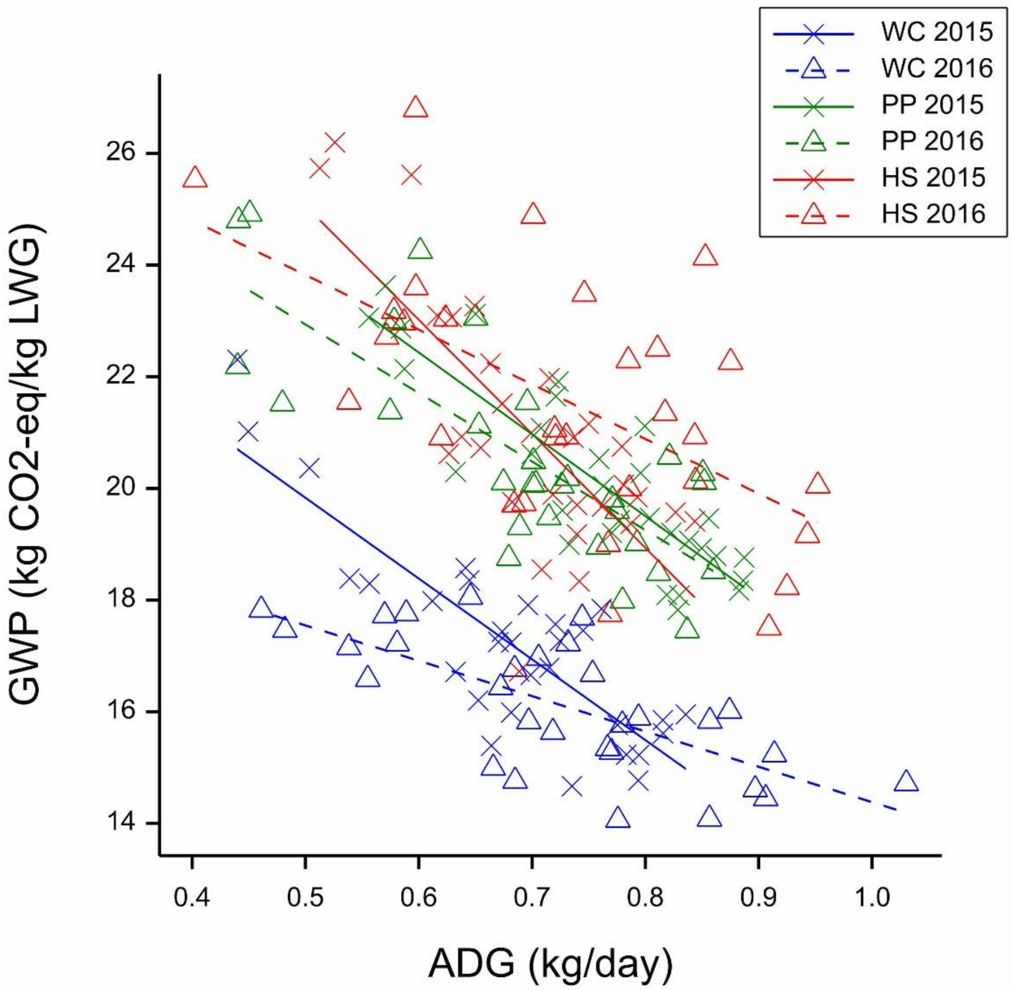


Figure A5.2. Relationship between individual animals' carbon footprints (as GWP per kg LWG) and average daily gains (ADG) while in the finishing enterprise. PP: permanent pasture; WC: white clover mix; HS: high sugar grass.

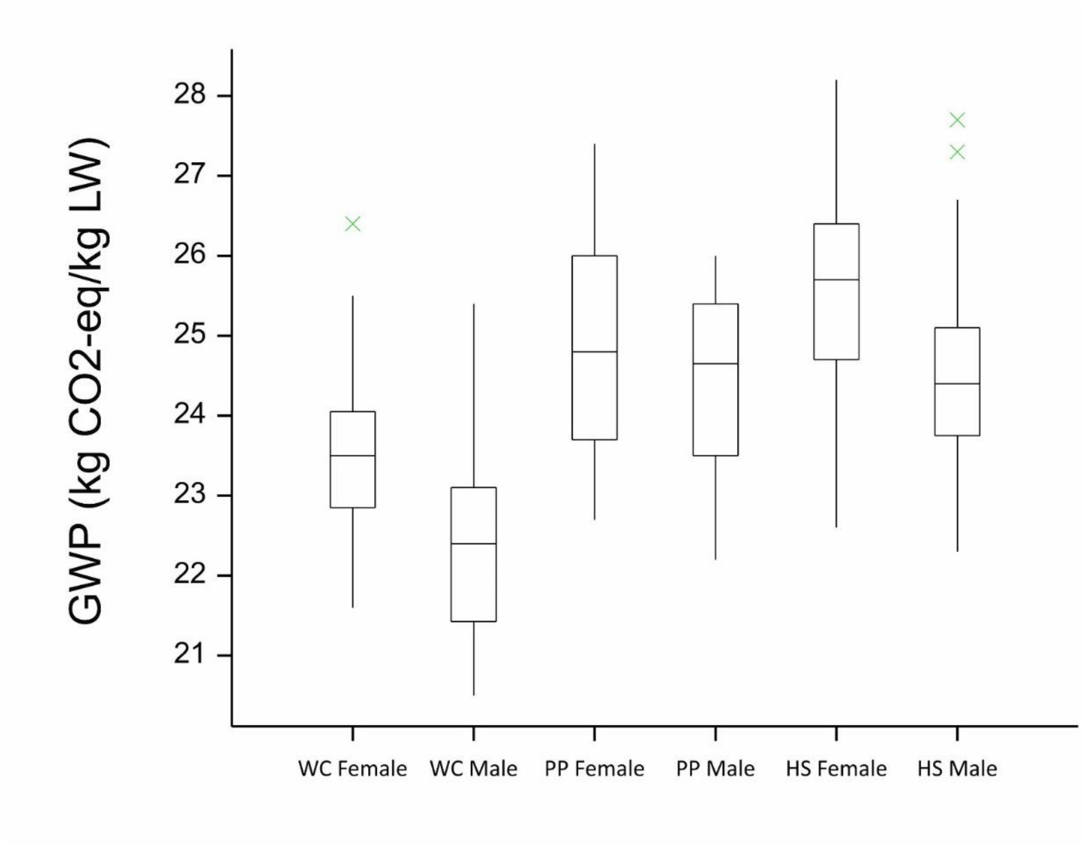


Figure A5.3. Distributions of Carbon footprints across farmlets and sexes. Outliers located further than 1.5 times the interquartile range beyond the quartiles are marked as crosses.

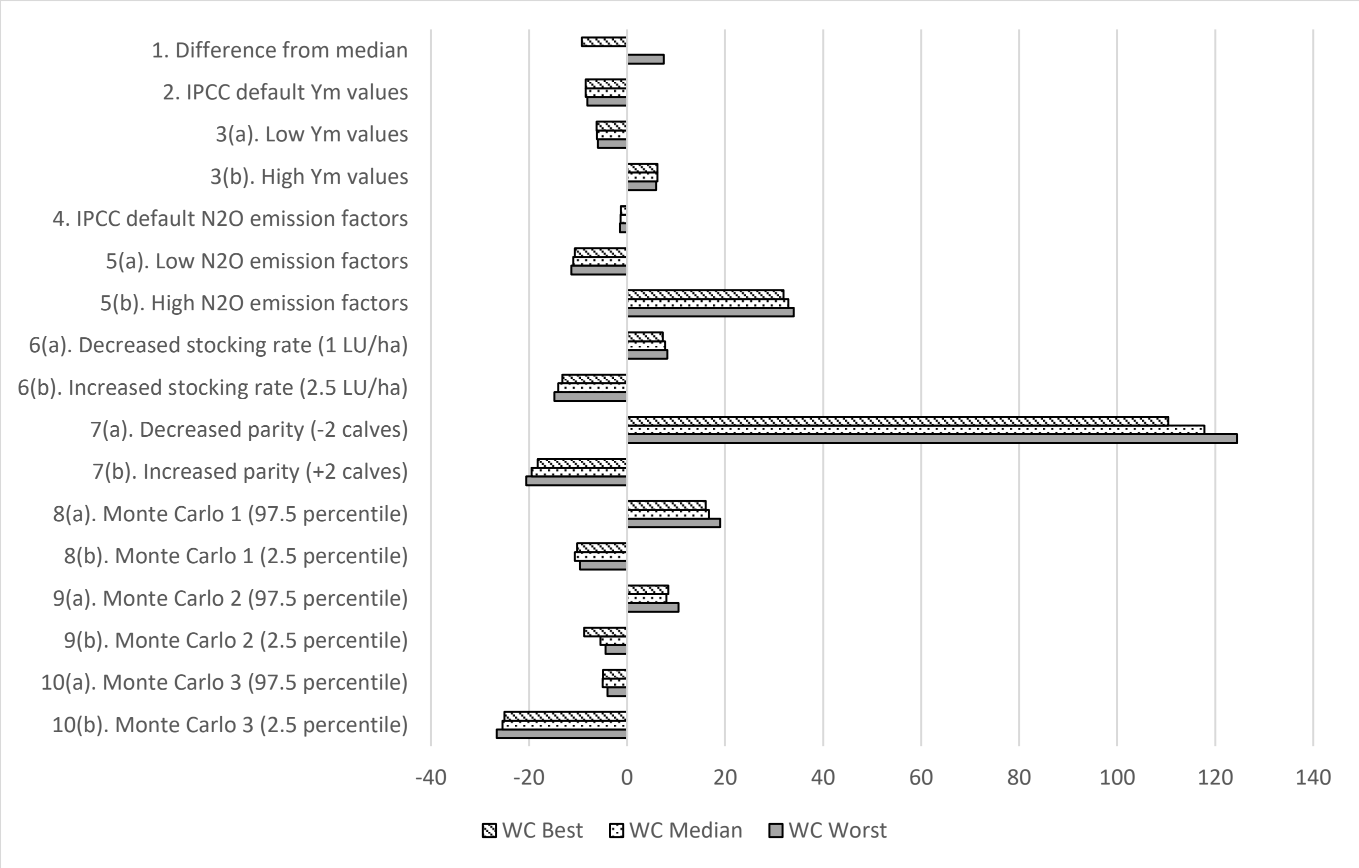


Figure A5.4. Results of scenario and sensitivity analyses carried out for the best (1st), median (15th) and worst (30th) animals on the white clover (WC) system based on their GWP rankings. All values are expressed as percentage differences from the baseline results for the same animals.

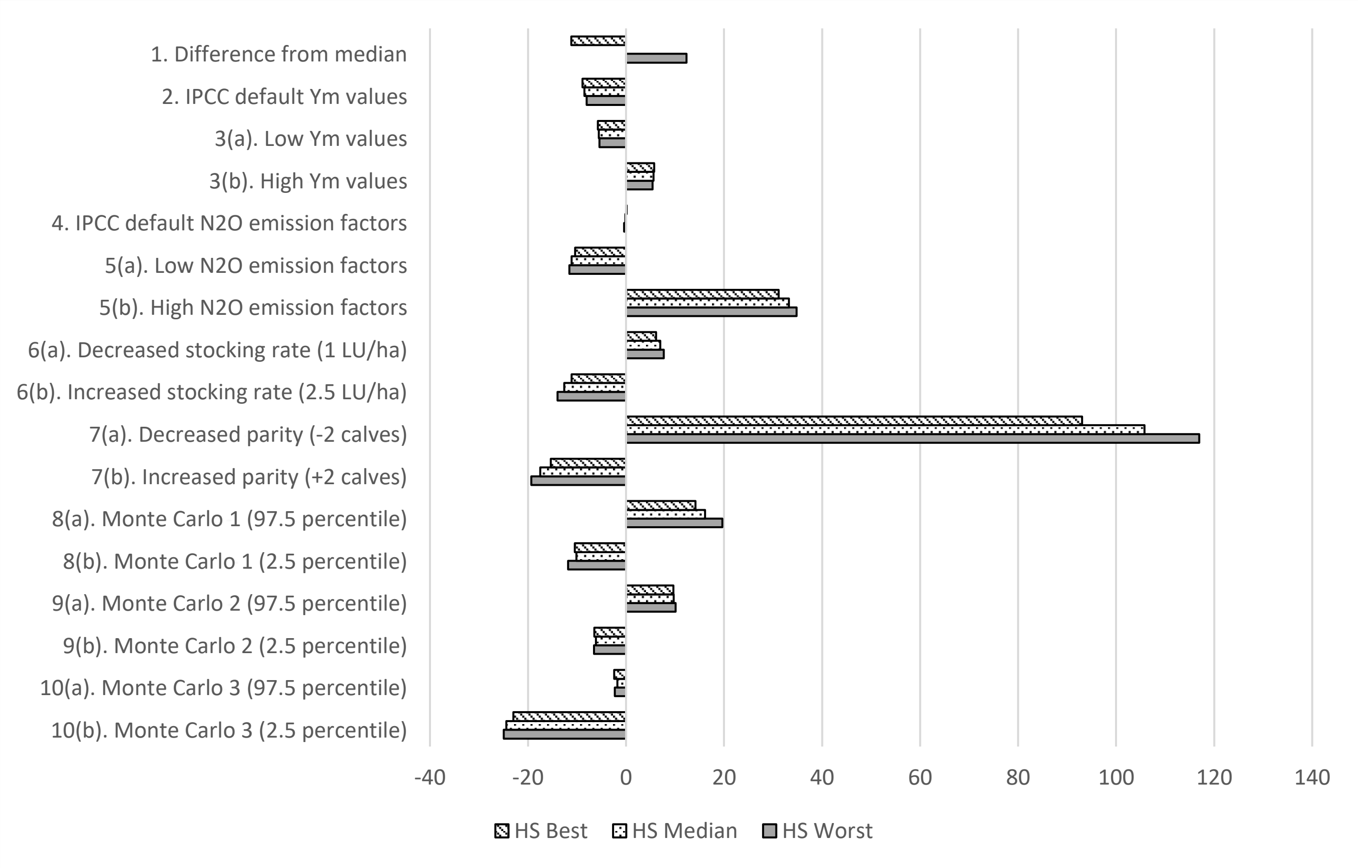


Figure A5.5. Results of scenario and sensitivity analyses carried out for the best (1st), median (15th) and worst (30th) animals on the high sugar grass (HS) system based on their GWP rankings. All values are expressed as percentage differences from the baseline results for the same animals.

Chapter 6 – Framework for life cycle assessment of livestock production systems to account for the nutritional quality of final products

6.1 Introduction

Studies employing life cycle assessment (LCA) of the agri-food sector, such as those produced in Chapters 2, 3 and 5, typically estimate pollution-production ratios as their primary outputs—for example kg CO_2-eq per unit of food produced—whereby systems represented by lower scores are judged to be socially more desirable. In the context of livestock production systems, denominators depicting the quantity of production, or the functional unit, generally takes the form of the mass of output, such as 1 kg of liveweight, cold carcase weight or deboned meat (de Vries and de Boer, 2010). While this traditional approach provides a useful means of inter-comparisons between different farming strategies, the resultant indicators are not a holistic representation of the real function of the final product, in this case meat, as a source of human nutrition. In other words, mass-based outputs of LCA studies implicitly bear an extra layer of uncertainty, especially when the same mass of food can have dissimilar values for society.

Recent research has begun to address this issue by means of dietary comparisons. Coelho et al. (2016), for example, examined the environmental impacts of hypothetical human diets with elevated omega-3 polyunsaturated fatty acid (PUFA) intake, made possible by adjusting livestock feeds to promote a higher omega-3 content in animal tissues, using the functional unit of "daily dietary ingestion per person". Society-wide dietary shifts, however, require drastic changes in supply chain structure as well as consumers' opinions, and therefore can only be achieved over a long period of time (Smil, 2000). More importantly, as any human diet is composed of a large number of food groups originating from multiple farms, implications of these studies on agricultural systems producing each commodity are not immediately clear. The latter problem is further exacerbated by the fact that a change in farming methods, however minor, often disrupts the flow of nutrients within the production environment and consequently leads to knock-on effects on chemical compositions of the end products, and ultimately their nutritional value to humans. This, in turn, poses a question about the assumption behind the majority of dietary comparison studies (and others adopting mass-based functional units) that all products are qualitatively homogenous. In order to draw short to medium-term recommendations for commercial agricultural producers to improve their environmental performance, it is therefore necessary to establish an LCA research framework which accounts for nutritional compositions of individual food groups that are produced under multiple production systems.

This final chapter of the thesis aims to identify the current knowledge gap concerning nutritional quality of agricultural products within the LCA literature and demonstrate a way forward to reduce uncertainty associated with the choice of functional unit for livestock LCA. To achieve this aim, the remainder of the chapter is structured as a combination of a two-part review and a two-part quantitative case study, organised as follows. Section 6.2 provides a review of recent literature which examines human diets within the LCA context. Section 6.3 offers an overview of the effect of farming strategies on the nutritional value of final products, with fatty acid profiles as a case exemplar. Section 6.4 then develops two quantitative frameworks, both of which are based on existing methods of nutritional analysis in an LCA context but modified for comparisons of meat products specifically, and Section 6.5 applies the new approaches to beef, lamb, chicken and pork production systems in the UK. Finally, Section 6.6 will conclude the entire thesis, with discussions on practical barriers to reduce system-wide uncertainty and pathways to overcome these challenges.

6.2 Life cycle assessment and human diets

With the accumulation of food-based LCA studies, a series of review papers have been published over the past decade. Each of these works has a specific focus, for example, comparison across plant and animal-originated food products (Roy et al., 2009), amongst animal-based products (de Vries and de Boer, 2010) and more specifically beef (de Vries et al., 2015) and pig (McAuliffe et al., 2016) production systems. These reviews all highlight issues surrounding cross-comparability amongst LCA studies, particularly related to system boundaries and functional units. At the same time, literature in recent years has identified an equally crucial challenge for agricultural LCA, namely incorporating the impact of food on human nutrition into environmental evaluations. Heller and Keoleian (2003) were pioneers in acknowledging that food consumption patterns should be incorporated into the LCA framework when they recognised sustainability-limiting factors such as rapid conversion of prime farmland (economic), illegal farm operatives (social) and excessive depletion of topsoil (environmental) in the US food system. More recently, Heller et al. (2013) reviewed work carried out over 10 years since their 2003 publication and proposed key areas which require further investigation. The authors noted that considering food quality (e.g. energy, fat, protein, mineral and vitamin contents as well as their bioavailability and compositions) rather than solely addressing quantity (mass of food-types in different diets) is critical to improve understanding of the food-environment nexus.

Following the publication of Heller et al. (2013), a new generation of work has been carried out to further investigate the interlinkages between human diets and their environmental footprints. In this section, findings from these new studies, published between 2014 and 2017, are summarised. In order to select the most relevant research, papers containing the search terms “life cycle assessment” and “nutrition” were retrieved from Scopus and, after sorting by relevance, the first 200 returns were considered for inclusion. If a paper contained material leading to the examination of the role of human nutrition under the LCA framework, either at the whole-diet level or through functional unit manipulation, it was included in the final collection. These criteria resulted in 14 relevant studies.

Doran-Browne et al. (2015) applied the concept of nutrient density scores (NDS) to greenhouse gas (GHG) emissions arising from a range of agricultural products typically found in southeast Australian diets. The NDS of each product was determined according to the Nutrient Rich Food model (NRF9.3: Fulgoni et al. (2009)), whereby higher contents of nine encouraged nutrients (protein, fibre, vitamins A, C and E, calcium, iron, magnesium and potassium) are associated with a higher score, and three discouraged nutrients (saturated fat, sodium and added sugar) with a lower score. The quantity of each nutrient present in a product was first divided by its recommended daily intake (RDI), or daily allowances (RDA) for limited nutrients, to obtain the percentage of RDI satisfied by the product, and subsequently converted to a weighted score according to its relative importance with respect to energy value. The NDS for each product was then derived as the difference between the sum of these weighted scores associated with "positive" nutrients and the sum of the similar scores associated with "negative" nutrients. The food products assessed were beef (lean and untrimmed), lamb (lean and untrimmed), regular milk, reduced fat milk, wheat (*Triticum aestivum*) flour and canola (*Brassica napus*) oil. When using the standard mass metric (t CO_2-eq/t product), the authors found that wheat flour generated the lowest GHG emissions while milk and canola oil had similar levels of impacts. Meat products had the highest impacts, with the lean cuts having a higher CO_2-eq value than the untrimmed cuts. However, when the novel metric (t CO2-eq/NDS) was applied, the lean cuts had considerably lower environmental impacts than the untrimmed cuts, and the gap between non-meat products and lean meat was substantially narrowed. Similarly, both regular and low-fat milks were found to have considerably lower impacts than canola oil, whereas wheat flour still had the lowest impacts. Although not fully covering the whole life cycle of products and also stopping short of distinguishing between different compounds within each nutrient group, for example between PUFA and monounsaturated fatty acids (MUFA), the study proposes a useful technique for comparing different food groups based on their nutritional value.

Hallström et al. (2015) carried out a meta-analysis to investigate the environmental impacts of dietary change based on the findings of 14 carefully selected papers. The authors considered the effects of dietary change on both global warming potential (GWP) and land use, though only four of the 14 papers addressed the latter. Of the studies assessed, the majority (nine) used energy as the functional unit, four considered RDI, and one used the standard mass-based unit (grams of product included in a meal). Hallström et al. (2015) examined change from a reference diet to eight alternative diets: vegan; vegetarian; ruminant meat replaced with pork and poultry; meat partially replaced by plant-based food; meat partially replaced by dairy; meat partially replaced by mixed food; balanced energy intake; and "healthy" diet. Out of the eight treatment categories, changing to vegan or vegetarian diets demonstrated the greatest savings to both GWP and land use. Interestingly, multiple studies reviewed by Hallström et al. (2015) suggested that replacing ruminant meat with pork or poultry could reduce GWP by, in some instances, more than a switch to vegan diet, although this naturally depends on the non-meat (including meat alternative) components of the original diet. Mirroring Doran-Browne et al. (2015), the authors conclude that the functional unit plays a large part in the relative rankings of products and reiterate the stance of Heller et al. (2013) that further multidisciplinary studies are required to improve current evidence of links between the environment and human nutrition.

Heller and Keoleian (2015) considered the environmental implications of changing typical American eating patterns and associated effects on food wastage. The authors examined consequences of changing the average diet (with an assumed energy intake of 2,534 kcal/day) and the diet based on the current recommended energy intake (for moderate activity levels, 2,000 kcal/day), to isoenergetic diets based on advised food patterns. The shift to recommended intake patterns for both caloric scenarios required increases in fruit, vegetables, pulses, seafood and dairy products, and decreases in meat, eggs, nuts, oils, solid fats and added sugars. However, the 2,000 kcal scenario required decreases in grain consumption, while the 2,534 kcal scenario required increases. GWP was then estimated from per capita food consumption derived from the Loss-Adjusted Food Availability data series (USDA ERS, 2012), taking into account the likelihood of food wastage for each food group. Most notably, GHG emissions from meat, egg and poultry sectors decreased under both scenarios, while there were large increases in GHG emissions attributable to increased dairy consumption. Overall, the authors found that switching to recommended food patterns would increase GHG emissions per capita by over 150% for the 2,534 kcal scenario. On the other hand, a 2,000 kcal diet sourced from recommended food groups would result in a decrease of ~10% in GHG emissions per capita.

In an attempt to identify a functional unit suitable for capturing a wider measurement of the sustainability of food products, Masset et al. (2015) examined environmental footprints of foods and drinks representative of a typical French diet under a number of different impact categories. The authors defined sustainable food products as low emitting, affordable and of high nutrient quality, determined collectively by a single score that takes the value of 0, 1, 2 or 3. Nutritional quality of food products was determined in one scenario using the French SAIN, LIM method, whereby five nutrients (protein, fibre, calcium, vitamin C and iron) are encouraged while three (saturated fat, added sugar and sodium) are discouraged. If a food product obtained more than 97% of its energy from fat (as is the case for nuts and oils), then vitamin E, MUFA and α-linolenic acid content were also accounted for in the encouraged nutrient profile. A product's overall sustainability score was then derived by comparing its performances in the aforementioned three areas against median values; a product received a point if its GHG emissions and price were lower than the median, and if its nutritional score was higher than median. Masset et al. (2015) argued that mass and energy-based functional units are generally unhelpful in determining sustainable products. In particular, they demonstrated how functional unit manipulation can affect relative rankings across food products, with those of monogastric meat products and fruits/vegetables easily reversed between energy-based and nutrition-based computations of GHG emissions.

As briefly mentioned in Section 6.1, Coelho et al. (2016) investigated the effect of increasing omega-3 in French diets across a range of impact categories. Using a 15-day average French diet as the baseline, the authors created three alternatives based on the same food groups but with smaller portions (healthy diet); a diet without fish; and a vegetarian diet. For each scenario, the authors then generated a hypothetical scenario, named BBC after the *Bleu-Blanc-Coeur* initiative which promotes enrichment of omega-3 in foods, whereby omega-3 content of livestock products was enhanced through alternative feeding regimes with higher rations of (omega-3-rich; predominately α-linolenic acid) alfalfa (*Medicago sativa*), sunflower (*Helianthus annuus*) meal and linseed (*Linum usitatissimum*). While the vegetarian diet performed most favourably across all but one impact category (GWP, acidification, eutrophication, land occupation and biotic natural resource-depletion species, with the exception of cumulative energy demand), *Bleu-Blanc-Coeur* generally generated lower burdens compared to the equivalent diet with typically lower omega-3 content. This result suggests that enhancement of nutritionally beneficial omega-3 fatty acids has the potential to simultaneously improve the environmental performance of supply chain, although economic feasibility of such a strategy would also have to be tested before a wider commercial roll-out.

Hess et al. (2016) considered the impact of starchy carbohydrate dietary shifts in the UK, from British potatoes (*Solanum tuberosum*) to Indian basmati rice (*Oryza sativa*) or Italian pasta, on GHG emissions and blue water scarcity footprint. Acknowledging that water contents of potatoes, rice and pasta are heterogeneous, the authors adopted a standardised functional unit for comparison: an average portion based on net weight of the product as packed. The size of a single portion was defined as 175 g for potatoes and 75 g for pasta and rice. Across the three products, basmati rice had higher GHG emissions and blue water scarcity footprint than the others, due largely to anaerobic soil conditions leading to methane production, and to the fact that India is more water scarce than the UK and Italy. These results highlight how shifting dietary patterns (e.g. less potato consumption and more rice consumption) can displace environmental burdens from the consuming nation to the producing nation, particularly in terms of water scarcity.

Stylianou et al. (2016) developed the Combined Nutritional and Environmental Life Cycle

Assessment (CONE-LCA) method to empirically apply the conceptual framework devised by Heller et al. (2013). CONE-LCA is a hybrid approach in the sense that it utilises traditional midpoint LCA modelling but adjusts output values for the endpoint based on the nutritional quality of food products. The ultimate output of this novel approach is the impact of diet change not only on midpoint environmental measures (GWP and respiratory effects via particulate matter in this particular case), but also the human health effects of these impact categories represented as disability-adjusted life years (DALY). The authors carried out a case study whereby an additional serving of milk (244 g) was added to three dietary scenarios: no changes to the rest of the diet; removing other food products with an equal caloric value (119 kcal); and removing an equal caloric quantity of sugar sweetened beverages. The authors used epidemiological data to assess milk's effects on human health, both positive and negative, as expressed by DALY. While adding a serving of milk to the diet increased both GWP and respiratory inorganics at the midpoint, milk consumption was found to be beneficial for long-term health, as reduced risk of colorectal cancer and stroke outweighed increased risk in prostate cancer in all scenarios. While Stylianou et al. (2016) acknowledge that uncertainty associated with endpoint impact assessments is considerable, their framework can be seen as a significant contribution to the methodological advancement of nutrition-based LCA.

Focusing on Dutch women aged between 31 and 50, Tyszler et al. (2016) used linear programming to examine the role of diet on a range of impact categories. This narrow sampling criterion was purposefully chosen to reflect greater requirements for iron, a common source of which is meat. The authors designed a simplified Dutch diet for this demographic group based on 207 individual products. A weighted scoring system was developed for overall impact assessment that accounts for GWP, fossil energy usage and land occupation. Using the typical diet as a reference point, a linear programming model was used to find a diet that was most similar to the reference diet but satisfied predefined nutritional or environmental constraints. The generated diets were classified as being nutritionally healthy and included: pescatarian; vegetarian; and vegan diets, as well as a diet unconstrained by food-groups, and a diet constrained by environmental performance (but unconstrained by nutritional composition). Tyszler et al. (2016) found that, for the patterns covered by this study, improved nutrition does not denote improved environmental performance. They also concluded that, while a general switch to vegetarian or vegan diets may have benefits in terms of certain impact categories, further research is required to assess the health implications of deficiencies of long chain PUFA such as omega-3 fatty acids.

Employing a similar framework to that adopted by Tyszler et al. (2016), Bälter et al. (2017) posed the question of whether a low GHG diet could provide the same nutritional values as a high GHG diet. Data on Swedish dietary patterns were collected through LifeGene, a planned long-term survey which aims to record lifestyle information at a 20-year interval. Participants were asked to record intake of all food products consumed at least once a month. The nutritional value of each diet was determined according to Nordic nutritional tables, while associated GHG emissions were sourced from published LCA literature. Where LCA studies omitted emissions attributed to post-primary production stages, the authors expanded the system boundary to include food processing, distribution and retail stages of the supply chain using Swedish data. Storage, cooking and waste management were excluded from the analysis. The study revealed that low GHG diets generally provide similar nutritional values to high GHG diets, with small differences largely driven by meat consumption. Similar to Treu et al. (2017), a study summarised later in the review, women were found to have lower dietary GHG emissions than men on average, also largely due to lower meat consumption. Bälter et al. (2017) concluded that a shift to diets higher in vegetables and lower in meat could reduce impacts to climate change.

Castañé and Antón (2017) compared the environmental impacts and nutritional quality of a Mediterranean diet representative of Catalonia and a recommended vegan diet designed specifically to address all nutrient requirements. The authors assessed nutritional quality based on the intake of nine encouraged and three discouraged nutrients (Fulgoni et al., 2009), and considered GWP and regionalised biodiversity impacts as impact categories. Fourteen daily menus were developed ranging from 1927 kcal to 2089 kcal of energy value. Protein, calcium and vitamin C were all higher in the Mediterranean diet than the vegan diet, but discouraged nutrients, particularly saturated fats, were also higher. Furthermore, the authors suggested that added sugar was higher in the Mediterranean diet as a result of meat and fish consumption, but mechanisms behind this causality were not elaborated on. Due in part to the selection of a nutritionally optimised vegan diet rather than actual consumption patterns amongst a vegan population, the nutritional quality of said diet was found to be higher than that of the Mediterranean diet, resulting in favourable environmental impacts per person per week for the vegan diet. However, the authors pointed out that deficiencies in micronutrients such as vitamin B12 and calcium under the vegan diet were not penalised under the proposed framework, reiterating the challenge of environmentally evaluating multiple diets when they may have different health implications.

In an examination of eating patterns amongst EU-27 countries, Notarnicola et al. (2017) employed LCA to quantify the environmental impacts of the diet consumed by a single average person over one year. Using 2010 as a reference year, consumption data were first aggregated into food groups such as meat, dairy products and vegetables. From each group, individual food items which represented the highest mass and economic values were selected for inclusion in the typical food basket. Unlike many LCA studies covering the agri-food sector, the authors set a cradle-to-grave system boundary, accounting for logistics, packaging, use and the end of life burdens in addition to production and processing. Data on downstream processes were largely sourced from international and European databases, including losses through food waste. A wide range of impact categories were considered, including GWP, eutrophication, acidification and land use, amongst others. Noternicola et al. (2017) suggested that meat and dairy products generated the highest burdens across most impact categories. Furthermore, under hypothetical scenarios with reduced beef and pork consumption, considerable reductions in environmental impacts were observed through replacement of meat with cereals. This approach, however, does not capture differences in product quality in terms of micro and macronutrients and, by extension, implications for human health.

Building upon the aforementioned NRF9.3 framework designed by Fulgoni et al. (2009), Saarinen et al. (2017) developed a novel nutrient index to specifically compare the overall quality of protein-rich foods. In addition to generic indices similar to Doran-Browne et al. (2015), novel formulae were devised to include MUFA, PUFA and vitamins B2 and B9, and at the time exclude nutrients that are not typically provided in abundance by protein-rich foods (e.g. magnesium and potassium). GWP was then estimated under both mass-based and nutritional score-based denominators. Nutrient contents of individual products as well as recommended intake values were sourced from public databases in Finland, while background LCA data were gathered through published literature. Twenty-nine food products ranging from cereals and pulses to dairy, meat and seafood were considered. Saarinen et al. (2017) demonstrated that the choice of functional unit can affect interpretation of results considerably. For example, beef had the largest GWP on a mass-based functional unit (100 g of product) but overtaken by cheese and lamb as the most burdensome food group when the functional unit was changed to the nutrient content included in 100 g of product. In general, animal-based products had higher environmental impacts than cereals and pulses regardless of the functional unit, although the authors did not consider contents of important micronutrients such as vitamin B12 in each food group.

Sonesson et al. (2017) developed a new approach to account for differences in protein quality between food products under the LCA framework. They established a new functional unit, protein quality index-adjusted mass, for each food product studied (bread, chicken breast, minced pork, minced beef, milk and pea soup) based on contents of the nine essential amino acids (EAA) in that particular product as well as in the overall diet. Protein quality index was formulated in such a manner that, if a particular EAA was deemed deficient in a given diet, food products with higher contents of the said EAA scored higher (and vice versa). The performance of each food product was then evaluated within the context of three diets: an average Swedish diet; a lacto-ovo vegetarian diet; and a low meat diet. Under the average Swedish diet, meat products, particularly beef, scored poorly and, as a result, were judged to have high environmental impacts. Conversely, under the low meat diet in which EAA such as leucine and lysine tend to be deficient, meat products scored favourably, and the results subsequently reversed. Through this example, the authors showcased the importance of considering the nutritional value of each food group in a wider picture of human dietary requirements.

Studying Chinese agri-food systems, Song et al. (2017) examined how dietary changes might abate climate change impacts and simultaneously improve public health. Due to a lack of available carbon footprint data in China, the authors used international databases to source GWP impacts for individual food products. Chinese consumption patterns were downloaded from a national health and nutritional survey and resulted in information from 11,160 respondents representing 5,253 households. A range of constraints were applied to an optimisation model, including maximum saturated fat intake and minimum protein and potassium intake, amongst others. The optimisation was then carried out to identify hypothetical diets which had the lowest carbon footprint while fulfilling nutrient requirements. The authors found that, in comparison to current typical diets, the optimisation process could decrease the carbon footprint of an average Chinese resident by 5–28%, primarily through reducing rice and meat consumption and increasing poultry consumption. The authors pointed out, however, that these conclusions were solely drawn from a single environmental indicator (GWP), and further work is required to provide a more holistic evaluation, such as the protein requirement for chicken production.

Treu et al. (2017) compared the carbon footprints and land use of conventional and organic diets in Germany. Diets representative of German consumption patterns were based on national statistics, and cross-database matching was carried out to pair diets with available environmental impact data. Food losses through supply chain wastage were also accounted for, resulting in impacts per required product rather than consumed quantities. The system boundary was set from the extraction of raw materials to the transport of products to a retail operator. Transportation from the retailer and cooking at home were excluded due to the lack of differences across both diets. The authors found that the carbon footprints of both diets were largely comparable. Animal based products accounted for 70% of GWP in both diets, as a higher quantity was included in the conventional diet but higher burdens per kg were associated with some products (e.g. poultry and pork) that constituted the organic diet. Interestingly, Treu et al. (2017) found that males, across both diets, tended to have higher GWP than females, due largely to higher meat consumption, as discussed above. For land use, the authors found that the organic diet requires about 40% more land than the conventional diet, further demonstrating the value of considering multiple metrics.

Considering the diets of 10,000 residents of Ontario, Canada, Veeramani et al. (2017)

examined the carbon footprints of typical consumption patterns based on single-day food purchases. Seven diets were assessed: vegan, vegetarian, pescatarian, omnivorous, and diets defined by the authors as “no red meat”, “no beef” and “no pork”. The functional unit to compare each diet was set as the annual energy intake by a typical person with a low activity level, or 837,436 kcal after adjusting for the female-male ratio. The two most common diets identified were omnivorous and no pork, under which the majority of impacts arose from beef production. Interestingly and in contrast to all other studies discussed above, the authors also considered a second functional unit formulated on protein requirements rather than energy requirements. Under this design, each diet had to provide 18.6 kg of protein annually to satisfy Canadian recommended values. The no pork diet remained as the highest contributor to GWP; however, the no red meat diet became the least polluting, while the vegan diet became the third most polluting (as opposed to least polluting when the functional unit was based on energy intake) due to the vast increases in food requirements to achieve the recommended protein intake.

6.3 Effects of farming systems on meat quality

Collectively, the studies reviewed in the previous section demonstrate the complex nature of interlinkages between human diets and the environment that need to be considered within nutrition-based LCA. Notwithstanding, it is increasingly recognised that mass-based assessments of agri-food systems are often inadequate at capturing the complexities of both food production (Martínez-Blanco et al., 2011) and wider supply chains (Schau and Fet, 2008) and, as a result, nutrition is rapidly becoming a key aspect of food LCA studies (Nemecek et al., 2016). In particular, findings by Sonesson et al. (2017) offer an important insight that the shift to quality-based functional units can dramatically alter the resultant environmental footprints of individual livestock products. As discussed, however, these efforts have mostly been confined to diet-level analyses, with few studies investigating the impact of altered functional units in a single-commodity setting.

Using fatty acid profiles of meat products as an example, this section summarises the current state of knowledge concerning how farm management affects the nutritional value of the final product. Fatty acid profiles were selected as a case exemplar because of their complicated role in human nutrition. As discussed in Section 6.2, it is generally accepted that omnivorous diets have larger environmental footprints than vegetarian or vegan diets of equal energy value. Nevertheless, given the forecasted growth in demand for meat into the future (FAO, 2015b), it would be prudent to identify methods of livestock production that offer the best balance between food security, natural environment and human health (Eisler et al., 2014).

Meat consumption, particularly that of red and processed meat, the latter of which often contains high levels of salt and sulphites, is commonly associated with an increased risk of cardiovascular disease (CVD) (Daviglus et al., 2017). With red meat being low in total fat (typically <5%), the causality appears to be driven by high proportions of short chain saturated fatty acids (SFAs), particularly C12:0 (lauric acid), C14:0 (myristic acid) and C16:0 (palmitic acid) (Micha and Mozaffarian, 2010), together with omega-6:omega-3 (ω-6:ω-3) ratios as high as 15:1 (Warren et al., 2008a). This, in turn, is perceived as a contributor to "unhealthy" Western diets with typical ω-6:ω-3 ratios in excess of 12:1, while the medically recommended ratio is around 3:1 (Simopoulos, 2006). When ruminant animals are finished on grass and clovers, however, their meat tends to have lower quantities of C16:0, higher quantities of C18:0 (stearic acid) as well as ω-6:ω-3 ratios of 2:1 or lower (Warren et al., 2008) and, contrary to prevalent perception, result in reduced risks of CVD and other inflammatory-driven diseases when consumed in moderation (Simopoulos, 2006). As a result, a growing body of studies indicate that advice on dietary restrictions (especially complete removal) of lean red meat may, in fact, be counterproductive to prevention of non-communicable disease (Binnie et al., 2014). Such considerable differences in health implications between meat products produced from forage-based and cereal-based feeds make the fatty acid profile of meat an ideal case to investigate the effect of accounting for human nutritional aspects of agricultural production systems in the environmental assessment framework.

Amongst various classes of fatty acids, omega-3 PUFA have particularly various health benefits, such as prevention of CVD and rheumatoid arthritis, as well as improvements to brain function and mental stability (Ruxton et al., 2004). While omega-3 has traditionally been considered beneficial only when maintained in a suitable ratio with omega-6 (Simopoulos, 2006), some research has subsequently challenged this theory, suggesting that the benefit of omega-3 should be considered solely in terms of total intake (Stanley et al., 2007). Importantly, the omega-3 content of meat products is known to be manipulated through livestock feeding strategies (Dewhurst et al., 2006, McAfee et al., 2010); in other words, a change in on-farm practice will likely have direct impacts on LCA results when the functional unit is nutrition-based.

To date, several reviews of the literature have been conducted on the relationship between farming systems and meat quality across different livestock species. Prior to quantitative case studies in Section 6.4, these articles will be summarised here. For the purpose of initial screening, papers containing the keywords "meat quality", "diet" and "review" were requested on Scopus without any restriction on their publication years. Similarly, to Section 6.2, resulting documents were then sorted according to relevance and the first 200 papers were considered for inclusion. From this pool, all abstracts were examined and studies reporting the effect of either diets or production systems on meat fatty acid profiles were shortlisted. Papers focused solely on novel and unconventional feeding strategies such as inclusion of tannins (Morales and Ungerfeld, 2015) or microalgae (Madeira et al., 2017) were excluded. Furthermore, selection was limited to beef, lamb, chicken, and pork—the four most commonly produced meats globally (OECD/FAO, 2017)—and therefore work on other meat (e.g., rabbit: Dalle Zotte and Szendrő (2011)) was also excluded. Based on these criteria, nine papers were selected to collectively provide an overview of farm management factors that influence fatty acid profiles. The first five papers below primarily review works on white meat (defined here as poultry and pork), while the last four cover red meat.

D'Arrigo et al. (2011) reviewed a range of fresh and processed meat products with an aim to identify functional foods, or foods which not only provide basic nutrition but also risk prevention from certain types of non-communicable diseases. The authors acknowledge that improving omega-3 compositions in the human diet is one of the main premises behind the functional food paradigm, with the adjustment of livestock feed being a key area of potential. For example, Enser et al. (1996) compared fatty acid profiles of beef, lamb and pork purchased from English retailers. Although pork had the highest PUFA:SFA ratio amongst the three products due to high levels of C18:2 omega-6 (linoleic acid), this also resulted in an undesirably high ω-6:ω-3 ratio of 7; whereas, the corresponding ratios for beef and lamb were 2 and 1, respectively. While chicken meat was not analysed as part of this study, its value has subsequently been shown to be comparable (7.6) to that of pork (Lee et al., 2012).

In a review on meat quality, Wood et al. (2004) summarised possible methods to increase omega-3 across pork, beef and lamb systems, e.g. through dietary supplementation using linseed. Supplementation for pigs has shown varying responses, with some studies reporting no adverse effects on meat composition (Enser et al., 2000) while others suggesting that feeding strategies which elevate C18:3 (α-linolenic acid) reduce palatability, particularly when interventional treatments such as salt injection are carried out (Myer et al., 1992).

Employing a systematic review approach, Corino et al. (2014) examined the effect of dietary linseed on the nutritional quality of pork and pork products. The authors considered the fatty acid profiles of 1006 pigs reported in 24 published papers and found positive effects of linseed supplementation to intramuscular fat and adipose tissue. In addition, a positive correlation between dietary treatment and both α-linolenic acid and C20:5 (eicosapentaenoic acid; EPA) was noted. While the evidence suggests such supplementation to be largely beneficial, not least due to economic feasibility, the authors highlight an increased risk of rancidity due to the greater oxidation potential of elevated PUFA levels in the meat. As a way to address this issue, they showed that feeding the entire linseed, rather than oil extracts, could decrease oxidation rates and consequently improve the shelf-life, due to the high levels of antioxidants present in seeds.

Bogosavljević-Bošković et al. (2012) carried out a review of broiler rearing systems to investigate if production practices affected meat characteristics, such as chemical composition of the end-product. Although chicken meat has been shown to be a good source of omega-3 for humans (Sioen et al., 2006), Bogosavljević-Bošković et al. (2012) point out that there are conflicting viewpoints on the determining factors of chicken meat quality. For instance, Holcman et al. (2003) found that chicken meat produced from both indoor and outdoor EU-regulated fattening operations did not result in significantly different chemical compositions. In contrast, Husak (2007) found that organically reared chickens had higher levels of omega-3 than meat from free-range or conventional birds. Unfortunately, these products were obtained from either retailers or wholesalers, and, consequently, their feed ingredients were unknown. Ponte et al. (2004) used controlled trials to examine the effects of alfalfa supplementation on chicken meat. The authors found that, while the legumes improved meat quality, poultry demonstrated lower feed conversion ratios and reduced weight gain, suggesting that forages may not be an efficient feed source for broilers. A later study demonstrated, however, that this negative effect can be partially offset by providing exogenous enzymes to utilise fibre and non-structural polysaccharides (Lee et al., 2016).

Motivated by declining fish consumption trends in the UK, Rymer and Givens (2005) explored existing literature to determine how omega-3 fatty acids could be enriched in the human diet via poultry meat. The authors acknowledge that, while typical poultry diets produce meat low in omega-3 fatty acids, alternative diets enhanced with α-linolenic acid (typically sourced from linseed) or EPA and C22:6 (docosahexaenoic acid; DHA) (typically sourced from marine products) generally result in meat richer in long chain PUFA. Regarding different cuts of meat, dark chicken meat tends to be higher in α-linolenic acid than white meat, whereas the reverse is true for EPA and DHA due to higher levels of phospholipid fractions in white meat. Nevertheless, the authors point out that the typically low levels of total lipids in white meat result in comparable levels of EPA and DHA across both cuts of meats, and therefore chicken meat, white or brown, could be used as a vehicle to improve uptake of omega-3 in human diets. As Bogosavljević-Bošković et al. (2012) noted, however, increased levels of PUFA in meat reduces oxidative stability and consequently shortens shelf-life unless animals are adequately supplemented with dietary antioxidants such as vitamin E, as previously discussed with pork.

Although the conversion efficiency of dietary PUFA into meat is lower for ruminants than for monogastric animals due to biohydrogenation in the rumen (a rumen bacterial response to detoxify unsaturated fatty acid through saturation), basal diets for beef and lamb systems generally contain higher levels of omega-3; forage, the major component of a ruminant's diet, typically comprises 50–75% omega-3 (α-linolenic acid) and 6–20% omega-6 (linoleic acid; Dewhurst et al., 2003). In a review of fatty acid profiles of meat products, Wood et al. (2008) summarised results by Warren et al. (2008a), an examination of the effects of breed (Aberdeen Angus x Holstein-Friesian vs. Holstein-Friesian) and diet (grass silage vs. concentrates) on meat quality. The authors found that Holstein-Friesian steers had higher levels of PUFA and PUFA:SFA ratios than Aberdeen Angus steers because of higher proportions of phospholipids in the total lipids. Grass silage universally increased omega-3 in the meat, with concentrates (cereals) conversely increasing omega-6. However, silage-fed animals had a lower PUFA:SFA ratio than concentrate-fed animals, due largely to higher fat deposition. Warren et al. (2008a) also found that as finishing age increased from 14 months to 24 months, intramuscular fat levels increased, especially in grass-silage diets. As with pigs and poultry, increased PUFA had a negative effect on oxidative stability and shelf-life; as Warren et al. (2008b) reported, however, forage contains high levels of natural antioxidants (carotene and vitamin-E) which can inhibit this negative effect.

Reviews by both Scollan et al. (2006) and Howes et al. (2015) further explored nutritional strategies to enhance long chain PUFA in beef. Specifically, Scollan et al. (2006) considered the role of genetics in fatty acid composition of meat, such as the thyroglobulin gene that regulates fat marbling and mutations of myostatin that decrease intramuscular fat content and increase muscle mass at the same time. Motivated by health-conscious consumers, Howes et al. (2015) reviewed current literature to identify opportunities to enhance long chain fatty acids (PUFA) in lamb fattening systems. Notably, the authors considered how specific cultivars of herbs and legumes might affect fatty acid profiles; Ådnøy et al. (2005), for example, demonstrated that botanically diverse mountainous swards (classified as native mixed pastures) produced lamb meat with higher levels of PUFA than lowland lamb. Howes et al. (2015) hypothesised that such increases in PUFA could result from a decrease in biohydrogenation caused by endogenous plant factors of a diverse sward. Factors contributing to reduced biohydrogenation were separately reviewed by Lee (2014) and Buccioni et al. (2012); as an example, red clover (*Trifolium pratense*) facilitates the flow of PUFA to the duodenum and then deposition into meat and milk, through the action of the enzyme system polyphenol oxidase in the rumen.

Venkata Reddy et al. (2015) carried out a review of papers studying differences in meat quality between animal sexes (e.g. heifers and steers). The authors highlight that the hormonal status of cattle plays a significant role in fat and protein distribution within muscles. For example, and perhaps unsurprisingly, they assert that meat quality from heifers is much higher than bulls, largely due to increased fat deposition in heifers which results in improved water-holding capacity. Consistent with the finding by Ardiyanti et al. (2009) that allele C in heifers produced higher levels of MUFA and PUFA (as well as lower levels of SFA), consumer panels have also demonstrated a preference for heifer beef over steer beef. More generally, feeding strategies that influence fatty acid profiles have implications on flavour and, consequently, preference; this point was exemplified by Sañudo et al. (2000), when British (grass-fed) and Spanish (concentrate-fed) lamb were offered to sensory panels in both countries. The panel in Britain preferred grass-fed lamb, whereas the Spanish panel preferred concentrate-fed lamb, reporting distaste for the “grassy” flavour. A similar tendency was observed by Larick and Turner (1990) for US sensory panels, who also preferred concentrate-fed beef over pasture-fed beef. Collectively, these results demonstrate that familiarity is a driving force behind consumers’ decision making.

6.4 Materials and methods

6.4.1 Omega-3 case study

In order to accurately connect the nutritional quality of meat products outlined above to the environmental footprints of farm management strategies under which they are produced, the following four steps need to be considered along the supply chain: (1) the environmental footprint per unit of farm-gate output (liveweight) under the studied farming strategy; (2) kill-out percentage of that particular animal; (3) meat yield from the carcase of that particular animal, and; (4) the nutrient content of meat from that particular animal. For the present case study, two functional units were selected based on the method of a preceding study (Marshall, 2001), namely the total mass of omega-3 PUFA and the combined mass of EPA and DHA, which together constitute a subgroup of omega-3 that are significantly more biologically active than shorter chain omega-3 and, therefore, do not need to compete with omega-6 for desaturase and elongase enzymes. The environmental footprints of different farming systems were estimated by combining studies that collectively cover the above four steps. Seven “treatments” or combinations of species and production systems commonly observed in the UK were identified: intensive cattle, extensive cattle, upland lamb, lowland lamb, conventional chicken, free-range chicken, and conventional pork. Feeding strategies reflected typical production practices for each system and therefore did not include supplementation of omega-3 rich feeds such as linseed. For each treatment, an LCA study and a meat science study reporting the fatty acid profiles were matched as closely as possible with respect to the underlying farming systems **(Table 6.1)**, and the global warming potential (GWP) was derived under each functional unit. GWP based on a standard mass-based functional unit (kg deboned meat) is also reported for methodological comparison.

Data pertaining to beef-related emissions were sourced from Audsley and Wilkinson (2014), of which dairy beef systems (slaughtered at 13 months) and suckler beef systems (18-19 months) were judged to be the most comparable, respectively, to the concentrate-fed beef and the silage-fed beef examined in Warren et al. (2008a). For fatty acid profiles, data from Holstein-Friesian cattle (on two feeding regimes) slaughtered at 19 months were adopted. As Audsley and Wilkinson (2014) utilise carcase weight as a functional unit, meat yield was estimated using the guidelines by van Leeuwen (2014a), which suggest the combined wastage rate (bone/fat/drip loss) of 13.0%.

Lamb production in the UK is typically carried out on both lowland and upland. To examine differences arising from these contrasting production environments, carbon footprints associated with both systems were sourced from Jones et al. (2014). As the functional unit adopted by the authors was 1 kg liveweight, GWP was first converted to represent 1 kg carcase weight (using the kill-out coefficient of 47.4%) and then to 1 kg edible meat (using the combined wastage rate of 12.2%), both based on van Leeuwen (2014b). Fatty acid profiles were sourced from Whittington et al. (2006), who conducted meat analysis of Suffolk lambs produced under lowland and upland systems.

GWP arising from broiler production was obtained from Leinonen et al. (2012), which employed a functional unit of expected weight of edible meat. As a result, no manipulations were made to derive the meat yield. Fatty acid composition was taken from Givens et al. (2011), who used whole cooked chickens for their meat analysis. Although cooked meat could potentially lose a portion of PUFA content as a consequence of oxidation, recent research has demonstrated that these losses are likely to be minimal (Douny et al., 2015).

LCA data for pork production was sourced from Audsley and Wilkinson (2014), whose study of typical pig production systems in the UK was carried out with the functional unit based on carcase weight. Meat yield was obtained from Marcoux et al. (2007), whereby 53.9% of a carcase is reported to be lean meat. Meat data were taken from Enser et al. (1996), who examined the fatty acid composition of typical pork cuts available at UK retailers. As the feeding regime of the animals used in the meat analysis was unknown, it was assumed that the cuts represent conventional (intensive) farming systems.

6.4.2 Nutrient index case study

While the approach described in Section 6.4.1 offers a useful framework for LCA when the research question primarily concerns a single nutrient, these functional units do not necessarily represent the overall value of the product associated with human nutrition. One way to address this issue is through the use of a nutrient index, a scalar value to combine information on multiple nutrients, both beneficial and detrimental to human health. For this case study, the following four variants of the formulae originally developed in a Finnish study by Saarinen et al. (2017) were adopted and applied to the seven livestock systems studied in Section 6.4.1: $UKNI_{prot}7$ and $UKNI_{prot}10$ based on $FNI_{prot}7$, and $UKNI_{prot}7\text{-}2$ and $UKNI_{prot}10\text{-}2$ based on $FNI_{prot}7\text{-}2$. The first group simply rewards foods with higher contents of desirable nutrients (protein, MUFA, EPA + DHA, calcium, iron, riboflavin, folate and, additionally for $UKNI_{prot}10$, vitamin B12, selenium, zinc), while the second group also penalises those with higher contents of undesirable nutrients (SFA and sodium). Only EPA and DHA were considered amongst PUFA so as to ensure their bioavailability (Section 6.4.1). Vitamin B12, selenium and zinc, which did not form part of the original indices, were added to the alternative specifications as meat is particularly rich in these micronutrients (Castañé and Antón, 2017). All four indices are expressed as % RDI per 100g, indicating the proportion of RDI satisfied across all nutrients, minus penalty where applicable, by the said amount of product.

RDI and RDA values were sourced from the British Nutrition Foundation (BNF, 2016), as averages between female and male. Where UK-specific recommendations were unknown or unspecified, as was the case with MUFA, values from Saarinen et al. (2017) were adopted. Nutritional compositions of (uncooked) meats were sourced from McCance and Widdowson (2015), except for fatty acid profiles carried over from the first case study **(Table 6.1)**. GWP estimates were also taken from the first case study and, contrary to Saarinen et al. (2017), excluded the cooking process to match nutritional data. Based on best available evidence, it was assumed that protein and micronutrient contents were unaffected by production systems for the same species (Scollan et al., 2006).

6.5 Results and discussion

6.5.1 Omega-3 case study

Table 6.2 provides a breakdown of fatty acid profiles adopted for the seven treatments. Mirroring the results from other studies, considerable differences were found between animals fed concentrates and forages, with more extensive systems generally producing more favourable profiles. Interestingly, the omega-3 content of free-range chickens was found to be lower than that of conventionally reared chickens. For many consumers who believe that free-range or organic meat products are healthier (Van Loo et al., 2010), this result may be unexpected. However, since the study by Givens et al. (2011) was based on meat purchased from supermarkets, the diets of chickens are unknown and, as a consequence, the reasons behind the PUFA differences cannot be completely ascertained, as in these systems diets will be based on least cost-rations and so may contain high level of omega-3.

GWP implications derived under the new functional units were profoundly different compared to the standard LCA results, particularly for beef and sheep systems **(Table 6.3)**. For example, concentrate-fed cattle produced approximately half the emissions of pasture fed cattle under the standard mass-based approach. When the omega-3 content of meat is considered, however, these results reversed and the concentrate-based system produced more than double the emissions of the pasture-based beef system. This difference was further exacerbated when only the most bioactive omega-3 fatty acids (EPA and DHA) were included. Between the two lamb systems, while the upland system had a marginally higher GWP, it also produced meat with a marginally higher omega-3 content, resulting in a minimal difference when the novel functional units were applied. Differences between free-range and broiler chickens were less pronounced because neither GWP nor omega-3 contents differ as substantially as cattle and lamb systems. Nonetheless, the higher levels of total omega-3 and EPA + DHA contained in intensively reared chickens increased the GWP gap between the two systems.

Across species, pig production was shown to be most affected when the functional unit was changed from mass-based to quality-based. While the new method did not alter the relative rankings between species, the discrepancy between red meat systems and white meat systems was considerably narrowed, challenging the view to stringently regulate ruminant production on the basis that it is far more harmful to society than monogastric production (Springmann et al., 2016). It could be argued that omega-3 should be sourced from alternative food groups such as oily fish and seafood, which are generally known to have higher contents of EPA and DHA than either white meat or red meat. Nonetheless, low consumption of these items in many societies suggests that, at least in short to medium terms, it is important to evaluate environmental impacts associated with production of all food types based on their nutritional values. More importantly, the current approach could be applied to any number of nutrients, so as to draw information not reflected when the mass of product is used as a sole reference to the value of food.

Finally, it is worthwhile reiterating that, in addition to containing higher levels of omega-3, forage-based production systems are also associated with lower ω-6:ω-3 ratios **(Table 6.2)**. Although quantifying this effect within the LCA framework is not straightforward and was therefore judged to be beyond the scope of the present case study, these systems are likely to result in further health benefits for humans than what is shown under the proposed functional units.

6.5.2 Nutrient index case study

When the seven systems were compared by the absolute level of nutrient scores, beef produced from forage-fed cattle was shown to be the most favourable product under all four index specifications (Table 4). All other systems, apart from intensive pork, performed comparably under $UKNI_{prot}7$, with pork scoring low due to lower contents of protein, MUFA and folate. Under $UKNI_{prot}7\text{-}2$ that also considers the two nutrients to be limited, beef and lamb produced the highest scores, while pork overtook free-range chicken due to its low SFA and Na. When the three additional nutrients (vitamin B12, Se and Zn) were further included (under $UKNI_{prot}10$ and $UKNI_{prot}10\text{-}2$), both beef production systems became notably more favourable than their counterparts from other species, owing to high concentrations of vitamin B12 and Zn. This finding is notable not only in the comparison between red meat and white meat but also between meat-based diets and plant-based diets, as vegan diets are often deficient in B12 and Zn, the latter more so amongst children (Gibson, 1994, Pawlak et al., 2013).

For computation of GWP, the mass-based functional unit **(Table 6.3)** was replaced with the four nutritional indices as denominators. As all nutrient scores are expressed as percentage, GWP values represent the environmental burdens associated with 1% of an average British person's nutrient intake in the form of that particular meat. It was found that the low mass-based GWP of chicken systems directly translated to low environmental impacts under both $UKNI_{prot}7$ and $UKNI_{prot}7\text{-}2$ **(Figure 6.1)**. The largely positive nutritional profiles of beef and, to a lesser extent, lamb, did not greatly alter the relative rankings under these index specifications. However, when vitamin B12, Se and Zn were introduced as nutrients to be encouraged, notable reversals in rankings were observed for cattle systems **(Figure 6.2)**. Concentrate beef generated the second lowest GWP only after intensive chicken under $UKNI_{prot}10$, and the lowest under $UKNI_{prot}10\text{-}2$. The performance of forage beef also improved, producing lower emissions than free-range chicken under $UKNI_{prot}10\text{-}2$. On the other hand, lamb systems consistently generated the highest burdens regardless of the index specifications, due to the significantly high mass-based GWP that were robust to different functional units. Nonetheless, the overall findings of this analysis question the appropriateness of comparing environmental performances of products on a mass basis—in a similar vein to the first case study.

6.5.3 Discussion

While recent work by Coelho et al. (2016), Heller et al. (2013) and Sonesson et al. (2017) provided useful strategies for assessing environmental implications of different diets on the whole, LCA literature remains short of methodologies to account for quality differences between individual foodstuffs produced under contrasting farming practices. The results from the above case studies suggest that the application of nutrition-based functional units in the single commodity setting has the potential to fill this research gap and offer better insight into economic-environmental trade-offs inherent by each production system and, by extension, on-farm strategies that should be promoted. Importantly, relative environmental performances amongst different agricultural systems reversed as new functional units were adopted, in particular between pasture-based and concentrate-based livestock systems, highlighting that the effect of farming methods on product quality should not be ignored in comparative studies. Nevertheless, improving nutritional values of meat (per GHG emissions) is only beneficial to the environment if it is accompanied by improved consumer awareness of differences in food quality, as exemplified by the *Bleu-Blanc-Coeur* initiative (Coelho et al., 2016), which subsequently leads to reduction in consumption of lower quality products. To this end, there is a clear need for further interdisciplinary work, including a scope for consequential LCA to account for wider socioeconomic impacts of dietary transitions as well as for end-point LCA to consider the ultimate impact of a product (and its quality) on human health. Even though a greater degree of uncertainty makes the latter a challenging task, work carried out by Sylianou et al. (2016) has paved the way to implement this concept. Finally, it should also be noted that GWP is one of many aspects of sustainability; in order to achieve a truly holistic comparison of livestock systems, a suite of metrics should be considered, including animal welfare (Edgar et al., 2013), land use (Wilkinson and Lee, 2018) and water quality (Leip et al., 2015), to name a few.

Needless to say, the validity of the approaches proposed in this chapter depends upon data reliability and the relative importance of the nutrients incorporated into the analysis. As already discussed, information from four steps along the supply chain (production, slaughtering, packing and consumption) needs to be linked together to enable the proposed framework and, while they should ideally be collected from a single agricultural system within a single region, such opportunities are rare and far between. The alternative method of collating separate works together poses the risk of inappropriately linked parameters, as carcase conformation, meat yield, chemical composition of meat and ultimately its human nutritional value are all strongly influenced by farming strategies that fundamentally regulate flow of nutrients—from soil to crops and then to animals.

As demonstrated here, it is possible to utilise existing datasets (from unrelated experiments carried out under similar environments) and create "hypothetical supply chains" that are sufficiently realistic for exploratory purposes. However, as the degree of uncertainty surrounding this approach cannot be specified, ideally a better way forward to overcome this issue would be to employ a whole supply chain approach (Orr et al., 2016), whereby actual products originating from different on-farm treatments are marked and tracked along the marketing process and used for quality evaluation and consumer trials. The finding from the present study, namely that nutritional quality rather than quantity is likely to play a key role in sustainable livestock production, warrants future studies in this area.

6.6 General conclusion

6.6.1 Summary of contribution to science

The ultimate goal of this thesis was to explore methods of reducing farm-level uncertainties of LCA studies targeting livestock production systems. In an effort to review and benchmark the method most commonly used in the preceding literature, Chapter 2 investigated differences in environmental footprints of pig production systems under three distinct levels of economic efficiencies using the concept of nationally representative farms **(Table 2.3)**. The most important finding from this study was that economic improvements, determined largely by feed conversion ratios (FCR), are likely to reduce environmental burdens **(Table 2.8)**. Although it is intuitive that improved FCRs will lead to reduced nutrient losses to nature, few studies have quantitatively tested this anecdotal evidence. In addition, the "food miles" concept was shown to be a misleading indicator of environmental footprints **(Figure 2.2)**. This possibility was suggested more than a decade ago (e.g. Dalgaard et al., 2007), but food producers, retailers and restauranteurs still use the idea as a method to attract consumers. As a consequence, it is valuable to disseminate transparent information of food systems and their associated environmental risks to the wider public. Regarding uncertainty, the commonly used Monte Carlo (MC) analysis was replicated to derive 95% confidence intervals around point estimates and to statistically compare footprints between different efficiency scenarios (Section 2.3.4). This approach is useful for capturing some uncertainties in LCA, such as potential variance of emission factors; however, it cannot capture many other sources of inventory uncertainties, such as biological growth rates of individual animals within a herd.

Motivated by this limitation of MC analysis in LCA, Chapter 3 set about developing a new approach to an existing modelling method (BSI, 2011, ISO, 2006). On the North Wyke Farm Platform (NWFP), all cattle are weighed every 2-4 weeks, providing a detailed understanding of how their energy requirements change over the course of a production cycle **(Figure 3.4)**. Moreover, it also creates the opportunity to investigate the role of growth performance on animal-level carbon footprints (CF). Using these data, in combination with a detailed inventory of material inputs and measured feed quality **(Table 3.1)**, a unique CF framework was devised, whereby CFs were calculated for each animal for each short period between two weighing events **(e.g. Figures 3.2 and 3.3)**. One of the main findings from this study was that average daily gain (ADG) was strongly negatively correlated with GWP **(Figure 3.5)**. This has positive implications for selective breeding and suggests that farms could improve their CFs as well as profitability if poorly performing animals were removed from the herd. However, this also raises a question over the accuracy of some existing studies which use average herd statistics to calculate burdens from a typical animal without considering variance in performance; **Table 3.5** shows that, when CFs are calculated under a "typical-animal" approach, the emissions intensity was notably lower than the arithmetic mean of the proposed individual animal approach. Collectively, the findings from Chapter 3 identified a layer of uncertainty which is often difficult to address due to data availability. Going forward, and where possible, practitioners should attempt to source more detailed information on livestock performance, whether at the farm-level, regional-level or national-level, in order to improve the reliability of their estimates.

Although Chapter 3 offered a methodological novelty in its own right, one empirical aspect which was lacking was the derivation of site-specific emission factors (EFs). The NWFP has a unique capability to integrate primary GHG measurements into animal production research (e.g. Cardenas et al., 2010, Cardenas et al., 2016, Horrocks et al., 2015, Misselbrook et al., 2014). Capitalising on these resources, Chapter 4 developed farm-level N_2O EFs to be used in subsequent, and improved, CF studies. For this purpose, a static-chamber experiment was set up on all three farmlets of the NWFP. Cattle urine and dung were collected from cattle and applied onto treated plots on the animals' respective fields **(Table 4.2)**. Two types of controls were also considered: one with nitrogen (N) (CON+N) and one without (CON-N). Gases were measured simultaneously on each farmlet throughout a grazing season. Once temporal variations in N_2O **(Figure 4.3)** were aggregated into total gaseous losses **(Table 4.3)**, EF_1 (% fertiliser N lost as N_2O) and EF_3 (% manure deposited on pasture lost as N_2O) were derived from emission values and identified N contents of applied urine and dung.

Building on from methods developed in Chapters 2 and 3, and data collected, processed and analysed in Chapter 4, Chapter 5 produced a novel CF study. Using the individual animal method devised in Chapter 3 for finishing systems, this study also incorporated the suckler herd to the model based on primary data collected from the breeding enterprise which provides weaned calves to the NWFP. As the most unique contribution of the study, site-specific EFs produced in Chapter 4 were used to estimate N_2O losses from fertiliser N and urine and dung, so as to eliminate reliance on IPCC (2006) default EFs with large confidence intervals. Interestingly, these farm-level EFs were not found to alter point estimates derived under the default EFs **(Figure 5.4)**, which is encouraging news for most CF studies of beef production systems in temperate grasslands. The inclusion of measured methane conversion values (Y_m), on the other hand, was found to affect results considerably more. Using Y_m values derived in a separate GreenFeed experiment at the NWFP resulted in approximately a 10% increase in CF **(Figure 5.4),** although care should be taken as feed used during the trial does not perfectly represent the annual nutritional intake by grazing animals. Another important finding in this study was that white clover (WC) performed considerably better environmentally than permanent pasture (PP) and high sugar grass monoculture (HS) across two production cycles (2015 and 2016), largely due to the reduction (in 2015) and omission (in 2016) of fertiliser N application. However, the benefits were more muted in Chapter 5 when compared to Chapter 3, explained by the inclusion of the suckler-herd, which accounted for around 60% of system-wide emissions. This suggests that more mitigation strategies should be levelled at the breeding phase of beef production. Indeed, improving parity and production efficiency of the breeding herd (via change in stocking rate) was found to generate a considerable reduction in CFs **(Figure 5.4)**, demonstrating the potential of improved herd management with regards to reduced environmental impacts.

All chapters preceding Chapter 6 were targeted at understanding and reducing farm-level uncertainties. However, one level of uncertainty which is rarely considered in LCA studies is that of product quality, an important driver of human health which remain uncertain under mass-based functional units. In order to fill this gap in the literature, the final chapter looked ahead towards a comprehensive framework for livestock LCA and proposed quantitative solutions to assess the real function of meat: human nutrition. When meat from four animal species were compared in terms of nutrients provided per unit of GHG emissions, beef was found to have a lower CF than chicken and pork **(Figure 6.2)** due to its denser nutritional values **(Tables 6.2 and 6.4)**. This suggests that, if humans consumed lean beef conforming to nutritional recommendations rather than today's typical portion sizes, much less portion would be required to achieve the same nutrient intake than other meats. It also suggests that existing comparisons of multiple foodstuffs on a per-mass basis may not reflect the whole picture of climate change impacts, especially when they have different levels of nutritional density. Nevertheless, reduction of excessive consumption patterns in the developed world should undoubtedly be a key target for lowering livestock GHG emissions.

6.6.2 Limitations of the present research

As outlined in the previous subsection, the majority of novelties in this thesis lie in methodological development and therefore are applicable regardless of physical and managerial properties of the farms for which future environmental evaluations are carried out. Empirically, however, the greatest limitation of this thesis, particularly in Chapters 3, 4 and 5, is that results reported here were generated from one study site (the NWFP) and do not capture inter-farm variation. As shown throughout this thesis, management practices greatly affect the environmental footprints of livestock systems. As data are still lacking to benchmark the NWFP's complete economic and environmental performances in relation to the population of commercial farms in England, the UK and beyond (Takahashi et al., 2018), the current research cannot draw definitive conclusions about the validity of quantitative results at a large geographical boundary.

As well as a lack of national-scale representativeness, a lack of economic analysis is also problematic in translating the findings from the current study into useful information for local farmers. For instance, the WC treatment at the NWFP continuously produced more favourable CFs than PP and HS. From this, it would be logical to deduce that WC is a socially more desirable system than PP or HS; however, this assertion would not be of interest to farmers if the cost of sward maintenance outweighs personal gains arising from the change in farming systems. Ideally, this thesis would have included an additional component to consider pathways to implement changes, in terms of both meat production and consumption, proposed across the six chapters. That said, cost-benefit analysis of various farm interventions for each of the NWFP's systems is ongoing, meaning this gap in knowledge will soon be rectified.

6.6.3 Future work

The NWFP is known as one of the most instrumented farms in the world, and collects a suite of information about grassland ruminant production systems. The current thesis utilised some of these data, such as detailed farm records and GHG measurements; however, there are a number of avenues which are yet to be explored. In LCA terms, perhaps the most beneficial function of the NWFP would be to use automated water quality measurements (Orr et al., 2016) to derive novel characterisation factors for eutrophication potential (EP). Currently, common freshwater EP factors are based on global phosphorus (P) fate models (Huijbregts et al., 2016), but water quality is known to vary considerably at the local level (DEFRA, 2014). Given the facility and expertise to measure P flows through soil and water at the NWFP, a future study should develop localised characterisation factors. This would be a considerable advancement to a very important aspect of LCA, in particular the trade-off between local and global pollution as discussed in Chapter 2, where current LCA literature lacks strong credibility.

In addition to the NWFP's resources, human nutritional work developed in Chapter 6 also warrants further investigation. For example, most studies assume that all diets require to supply the same level of nutrients. People tend to follow very different consumption patterns, however, both across nations and across individuals within nations. Therefore, future work could examine a range of actual, rather than recommended, dietary patterns based on best available data, and determine how individual food products would contribute in terms of diet-level nutritional requirements without incurring excessive environmental burdens. Similar work has already been carried out by Sonesson et al. (2017) in terms of protein quality via essential amino acids, but there is scope to look at wider nutrient groups such as various forms of fats, minerals and vitamins. Another aspect of nutritional LCA which holds potential to account for product quality is considering the addition of a single serving of a particular product to a diet and its subsequent epidemiological effects in endpoint modelling, as exemplified by Stylianou et al. (2016). Similar studies are currently lacking in the UK.

In conclusion, this thesis has taken a number of steps forward in identifying novel sources of uncertainty, most notably at the farm level. From the consideration of individual animals, site-specific EFs and the inclusion of meat nutrition, pathways are relatively straightforward to reduce said uncertainties; however, the vast sums of data required to do so severely restrict many LCA practitioners' options. In terms of environmental assessment as a whole, the current research sends a clear message that the single most important factor required to improve our understanding of livestock production systems lies with data collection. National bodies responsible for aggregating summary statistics should place more emphasis on intra-farm variations, such as those arising from differences in yield and daily weight gains, which could be achieved by carrying out only slightly more detailed farm surveys. When practitioners have a better understanding of uncertainties within and across farms, regions and nations, LCA comparisons will become scientifically more robust than simply comparing systems based on point estimates or range estimates derived from “default” Monte Carlo analysis, which the current thesis suggests may be flawed approaches. Finally, and looking ahead, the need for environmental interventions must be discussed together with their potential impacts on the availability of affordable food, particularly in developing countries (Eisler et al., 2014). International collaboration is the key for successful data collection and analysis in this area, and international networks such as the Global Farm Platform (GFP) (Gill et al., 2018), which the NWFP forms part of, hold opportunities of developing global databases with processes pertinent to farming practices from other instrumented farms all over the world. Detailed inventories from developing countries are not as prevalent as those from developed countries within Europe or North America. As a consequence, building upon our current understanding of agriculture’s relationship with the environment is critical to identify global strategies to avoid the biosphere’s degradation. Accepting the ethos that not-one-size-fits-all is critical in devising sustainable agricultural systems across the planet.

Table 6.1. Unit comparability between preceding works selected for the case study.

Species	System	GWP Study	GWP Unit[a]	Carcase Study	Carcase Unit	Omega-3 Study	Omega-3 Unit
Beef	Concentrate	Audsley and Wilkinson (2014)	7.9 kg CO2-eq/kg CW	van Leeuwen (2014a)	0.87 kg meat/kg CW	Warren et al. (2008a)	20 mg/100g meat
	Forage		15.9 kg CO2-eq/kg CW				97 mg/100g meat
Chicken	Intensive	Leinonen et al. (2012)	4.4 kg CO2-eq/kg MW	Leinonen et al. (2012)	Not required	Givens et al. (2011)	362 mg/100g meat
	Free range		5.1 kg CO2-eq/kg MW				214 mg/100g meat
Lamb	Lowland	Jones et al. (2014)	10.9 kg CO2-eq/kg LW	van Leeuwen (2014b)	0.88 kg meat/kg CW[b]	Whittington et al. (2006)	94 mg/100g meat
	Upland		12.9 kg CO2-eq/kg LW				103 mg/100g meat
Pork	Intensive	Audsley and Wilkinson (2014)	4.0 kg CO2-eq/kg CW	Marcoux et al. (2007)	0.54 kg meat/kg CW	Enser et al. (1996)	51 mg/100g meat

[a]CW: carcase weight; LW: liveweight; MW: meat weight
[b]Converted from LW based on the kill-out rate estimated by van Leeuwen (2014b)

Table 6.2. Summary of omega-3 and 6 fatty acid profiles reported in preceding works selected for the case study.

Species	System	Study	Omega-3 (mg/100 g meat)	DHA + EPA[a] (mg/100 g meat)	ω-6:ω-3[b]
Beef	Concentrate	Warren et al. (2008a)	20.3	3.4	14.4
	Forage		97.2	27.4	1.2
Chicken	Intensive	Givens et al. (2011)	362	17.6	5.5
	Free range		214	14.7	7.6
Lamb	Lowland	Whittington et al. (2006)	94.0	26.4	1.2
	Upland		103	31.7	1.5
Pork	Intensive	Enser et al. (1996)	51.3	14.8	7.4

[a]DHA: docosahexaenoic acid; EPA: eicosapentaenoic acid. They are a subgroup of omega-3 fatty acids that are the most biologically active and do not need to compete with omega-6 for enzymes.
[b]ω-6:ω-3: the mass ratio between omega-6 and omega-3 fatty acids

Table 6.3. Global warming potential (GWP) under different functional units.

Species	System	Mass-based GWP (kg CO_2-eq/kg meat)	Quality-based GWP (kg CO_2-eq/g omega-3)	Quality-based GWP (kg CO_2-eq/g EPA + DHA[a])
Beef	Concentrate	9.8[b]	48.0	288.1
	Forage	18.3[b]	18.5	67.7
Chicken	Intensive	4.4	1.2	25.1
	Free range	5.1	2.4	34.7
Lamb	Lowland	26.1[b]	28.7	99.2
	Upland	30.9[b]	30.0	98.9
Pork	Intensive	7.4[b]	14.4	50.3

[a]DHA: docosahexaenoic acid; EPA: eicosapentaenoic acid. These are a subgroup of omega-3 fatty acids that are the most biologically active and do not need to compete with omega-6 for enzymes.
[b]Recalculated from values reported by the authors for cross-study comparability

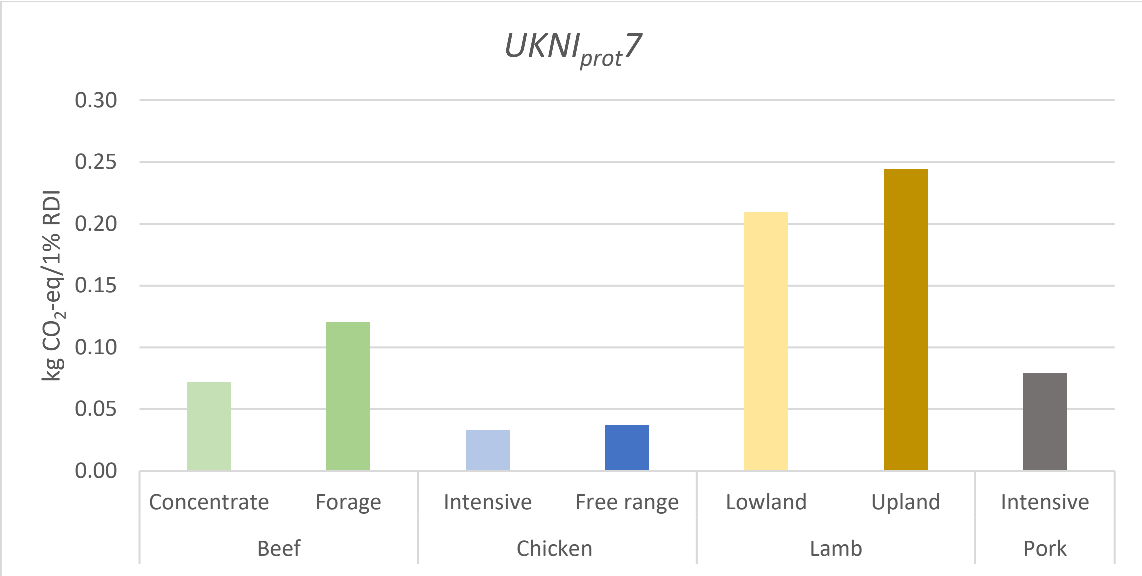
Table 6.4. Nutritional composition of each meat product (100g) considered.

Nutrient/index	Unit	RDI/RDA [a]	Beef		Chicken		Lamb		Pork
			Concentrate	Forage	Intensive	Free range	Lowland	Upland	Intensive
Protein	g/day	50.25	23.5	23.5	26.3	26.3	20	20	18.6
MUFA	g/day	37.5	1.1	1.6	3.7	5.4	1.3	1.1	0.9
EPA+DHA	mg/day	250	3.4	27.4	17.6	14.7	26.4	31.7	14.8
Ca	mg/day	700	5	5	11	11	12	12	10
Fe	mg/day	11.75	1.6	1.6	0.7	0.7	1.4	1.4	0.4
Riboflavin	mg/day	1.2	0.26	0.26	0.15	0.15	0.2	0.2	0.18
Folate	µg/day	200	16	16	9	9	6	6	1
Vitamin B12	µg/day	1.5	2	2	0	0	1	1	1
Se	µg/day	67.5	8	8	15	15	3	3	11
Zn	mg/day	8.25	4	4	1.5	1.5	2	2	1.3
Na [b]	g/day	6	0.07	0.07	0.08	0.08	0.07	0.07	0.053
SFA [b]	g/day	25	1.1	1.5	2.4	3.7	1.3	1.2	0.9
UKNIprot7	% RDI		13.6	15.2	13.4	13.9	12.4	12.7	9.4
UKNIprot7-2	% RDI		10.7	11.6	7.9	5.9	9.2	9.7	7.1
UKNIprot10	% RDI		28.9	30.0	13.4	13.8	18.2	18.4	16.4
UKNIprot10-2	% RDI		26.0	26.4	7.9	5.7	15.0	15.4	14.2

[a] Recommended daily intake/allowance based on BNF (2015) and Saarinen et al. (2017)

[b] Nutrients to be discouraged

A.



B.

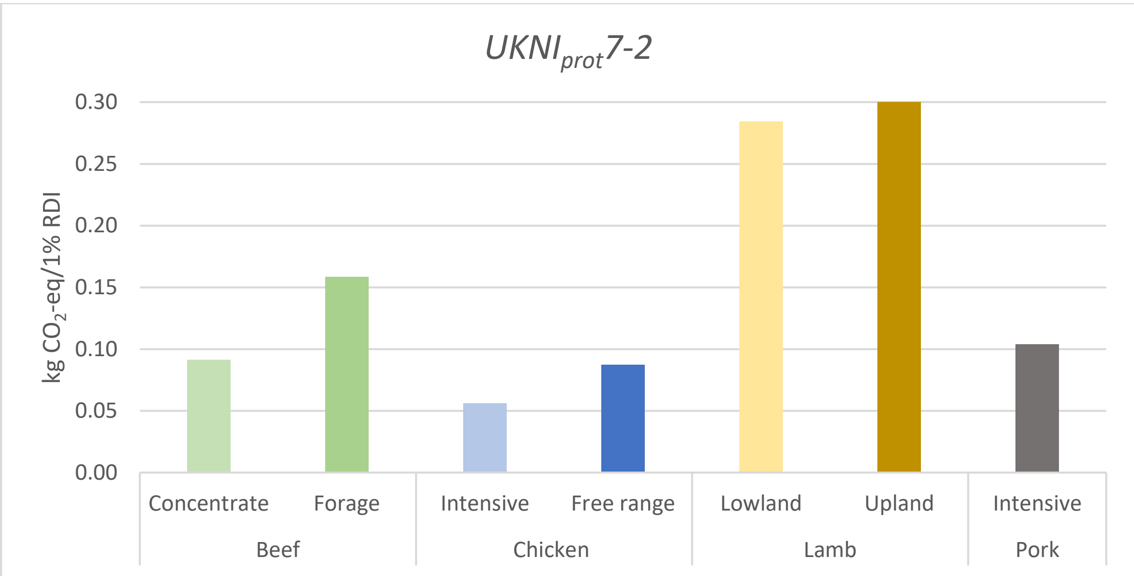
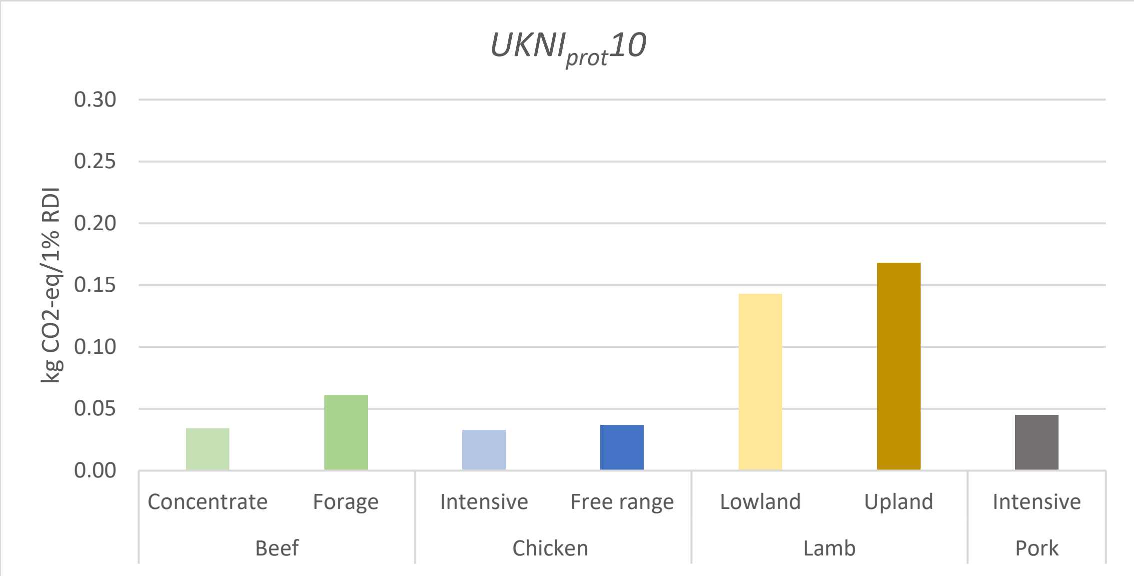


Figure 6.1. Global warming potential scaled to 1% of RDI under (A) *UKNI$_{prot}$7* and (B) *UKNI$_{prot}$7-2* specifications.

A.



B.

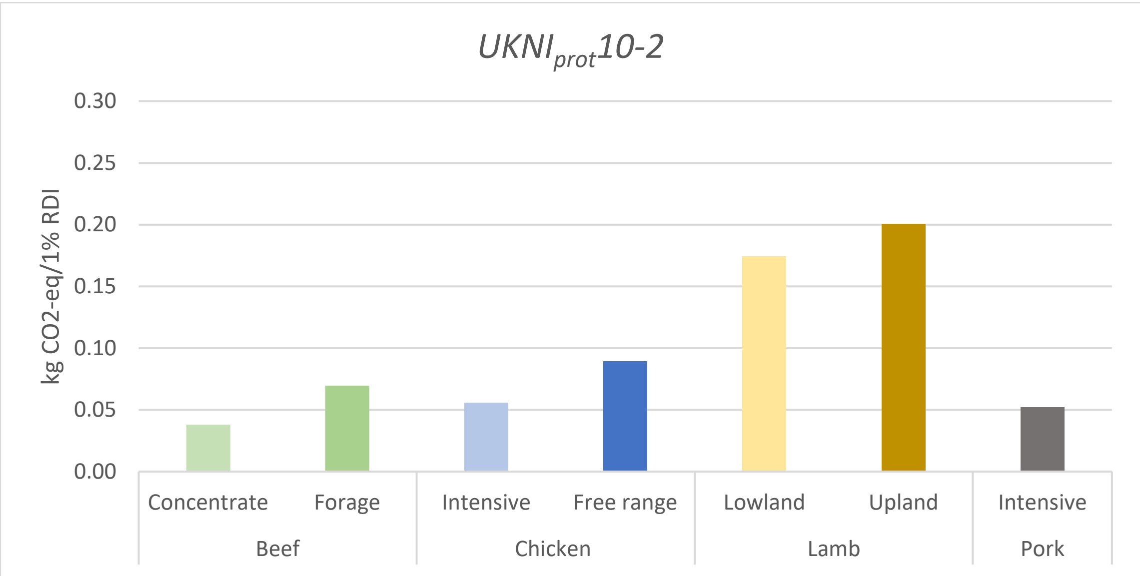


Figure 6.2. Global warming potential scaled to 1% of RDI under (A) *UKNI$_{prot}$10* and (B) *UKNI$_{prot}$10-2* specifications.

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