Modelling the environmental impacts of pig farming systems and the potential of nutritional solutions to mitigate them

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Abstract

The overall aim of this thesis was to model the environmental impacts of pig farming systems in Canada using Life Cycle Assessment (LCA), and to quantify the potential of nutritional solutions to reduce these. To achieve this, methodological challenges regarding co-product allocation, modelling uncertainty in agricultural LCA and how to formulate pig diets for environmental impact objectives needed to be resolved.

The options for co-product allocation in LCA studies of agricultural systems were evaluated and it was concluded that economic allocation was the methodology that could be adopted most consistently throughout the feed supply chain in livestock LCA models. A LCA model which quantified the environmental impact of Canadian pig farming systems, for multiple impact categories, was developed for the first time. A new approach to uncertainty analysis for the LCA of livestock systems using parallel Monte-Carlo simulations was also developed. The potential of including specific by-products from the human food and bio-ethanol supply chains in pig diets to reduce environmental impacts was investigated. Wheat shorts and bakery meal were found to reduce the environmental impact of the system in the scenarios tested. Further integration of diet formulation techniques with the LCA model allowed pig diets to be optimised explicitly to minimise environmental impact, while accounting for the effect of diets on nutrient excretion and the effect of energy density on feed intake in order to determine the optimum energy density of pig diets for different objectives. The potential effect of three environmental taxes, a carbon tax, and taxes on spreading N and P in manure respectively, on formulating pig diets and their implications for environmental impact were also modelled. The carbon tax was the only tax which consistently caused significant reductions in any of the impact categories tested in the LCA.

Overall, novel methodologies for modelling uncertainty in livestock LCA and formulating pig diets to minimise environmental impacts were developed. Using the latter, pig diets were formulated to reduce the environmental impacts of the production system for multiple impact categories simultaneously for the first time. These methods allowed the potential of dietary alterations to reduce the environmental impact of pig farming systems to be investigated systematically, and have wider applications for LCA modelling in livestock systems.

Declaration

This thesis has been composed by myself and has not been submitted as part of any previous application for a degree. All sources of information have been specifically acknowledged by means of referencing.

Stephen Mackenzie

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List of Abbreviations

ADG	Average Daily Gain
AP	Acidification Potential
BW	Body Weight
CW	Carcass Weight
C$	Canadian Dollars
DDGS	Dried Distillers Grains with Solubles
ECW	Expected Carcass Weight
Eq	Equivalent
EP	Eutrophication Potential
FAO	Food and Agriculture Organisation
FCR	Feed Conversion Ratio
FEP	Freshwater Eutrophication Potential
G:F	Gain to Feed ratio
G/F	Grower/Finisher
GWP	Global Warming Potential
GHG	Greenhouse Gas
IDF	International Dairy Federation
ISO	International Organisation for Standardisation
Kg	Kilogram
kWh	Kilowatt-hour
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LEAP	Livestock Environmental Assessment and Performance partnership
LW	Live Weight

LPG	Liquefied petroleum gas
MCDM	Multi-Criteria Decision Making
ME	Metabolisable Energy
MEP	Marine Eutrophication Potential
MJ	Megajoules
NE	Net Energy
NRE	Non-Renewable Energy use
NRRU	Non-Renewable Resource Use
OMAFRA	Ontario Ministry of Food and Rural Affairs
SEM	Standard Error of the Mean

Table of Contents

Chapter 1: Introduction **1**

1.1 The environmental impact of livestock systems 1

1.2 The environmental impacts of pig farming systems 1

1.2.1 Canadian pig farming systems 2

1.3 Life Cycle Assessment 3

1.3.1 Life Cycle Assessment of pig production systems 4

1.2.3 Reducing the environmental impacts of pig systems through dietary change 5

1.4 Methodological challenges for livestock LCA modelling 6

1.4.1 Co-product allocation 7

1.4.2 Modelling uncertainty 10

1.5 Thesis aims 11

Chapter 2: The need for co-product allocation in the Life Cycle Assessment of agricultural systems – is "biophysical" allocation progress? **12**

2.1 Abstract 12

2.2 Introduction 12

2.3 Co-product allocation and its use in agricultural LCA 14

2.4 Allocation using underlying physical relationships in an industrial setting 16

2.4.1 Example 1 – Allocation in the boron co-product system (Azapagica and Clift, 1999a) 16

2.5 "Biophysical" allocation in Agricultural LCA 18

2.5.1 Example 2 – allocation methodologies for co-products from dairy systems 21

2.5.2 Example 3 - manure as a "co-product" in layer systems 22

2.5.3 Example 4 – "The construction cost of plants" 24

2.6 Discussion and conclusions 26

Chapter 3: Accounting for uncertainty in the quantification of the environmental impacts of Canadian pig farming systems **29**

3.1 Abstract 29

3.2 Introduction 30

3.3 Materials and methods....30
3.3.1Model structure....30
3.3.2 Impact Assessment....31
3.3.3 Feed Ingredients....32
3.3.4 Farm Model....33
3.3.5 Manure model....36
3.3.6 Impact Sources....37
3.3.7 Sensitivity analysis....37
3.3.8 Uncertainty Analysis....38
3.4 Results....39
3.4.1 Impact analysis and comparison....39
3.4.2 Sensitivity analysis....47
3.5 Discussion....51
3.5.1 Uncertainty Analysis....51
3.5.2 Sensitivity analysis....53
3.5.3 Comparison with previous LCA studies....55
3.6 Conclusion....56
Chapter 4: Can the environmental impact of pig systems be reduced by utilising co-products as feed? 58
4.1 Abstract....58
4.2 Introduction....59
4.3 Materials and methods....60
4.3.1 The diets....60
4.3.2 Maximum Inclusion Levels....64
4.3.3 The LCA model....65
4.3.4 Feed Production....66
4.3.5 Farm model....68
4.3.6 Manure model....69

4.3.7 Environmental impact calculations 69

4.3.8 Uncertainty Analysis 70

4.4 Results and discussion 70

4.4.1 Experiment 1 70

4.4.2 Experiment 2 74

4.5 General Discussion 76

4.6 Conclusions 79

Chapter 5: Towards a methodology to formulate sustainable diets for livestock: accounting for environmental impact in diet formulation 80

5.1 Abstract 80

5.2 Introduction 81

5.3 Materials and Methods 82

5.3.1 The system under consideration 82

5.3.2 The LCA model 83

5.3.3 Feed production 84

5.3.4 Manure model 86

5.3.5 Farm performance 87

5.3.6 Quantifying environmental impacts 87

5.3.7 Diet formulation rules 88

5.3.8 Diets formulated 92

5.3.9 Dietary comparisons in the LCA model 93

5.4 Results 94

5.4.1 Diet composition 94

5.4.2 Environmental Impacts – Eastern Canada 100

5.4.3 Environmental Impacts - Western Canada 103

5.5 Discussion 106

5.5.1 Methodological Challenges 106

5.5.2 Formulation strategies for environmental impact reduction 109

5.5.3 Effectiveness of optimisation as a strategy to reduce environmental impact in pig systems 110
5.6 Conclusions 111
Chapter 6: The potential of taxes to reduce environmental impact of pig farming systems through altering the composition of pig diets 113
6.1 Abstract 113
6.2 Introduction 114
6.3 Materials and Methods 116
6.3.1 The system considered 116
6.2.2 Diet Formulation 117
6.3.3 Quantifying environmental impacts 119
6.3.4 Uncertainty Analysis in the LCA model 120
6.3.5 Taxation Levels 120
6.4 Results and discussion 125
6.4.1 Carbon tax 125
6.4.2 Nitrogen tax 130
6.4.3 Phosphorus tax 134
6.5 General discussion 139
6.6 Conclusions 141
Chapter 7: General Discussion 143
7.1 Modelling the environmental impacts of pig farming systems 143
7.1.1 The potential of nutritional solutions to reduce the environmental impact of the pig farming systems 144
7.2 Methodological issues for LCA modelling of Livestock Systems 147
7.2.1 Co-product allocation 147
7.2.2 Uncertainty analysis 150
7.2.3 Diet formulation for environmental impact objectives 151
7.3 Environmental impact characterisation 152
7.4 LCA software and modelling limitations 153

7.5 Utilising animal growth models in diet formulation 155

7.6 Sustainability modelling – wider issues for pig farming systems 156

7.7 The potential effect of environmental taxes on environmental impact from livestock systems 158

7.8 Scope for future research 159

References 161

Appendix A1: Composition of typical diets for pig systems in Eastern and Western Canada (Chapter 3) 187

Appendix A2: Composition of experimental diets in Chapter 4 192

Appendix B: Minor ingredient data sources 209

Appendix C: On farm energy use data 210

Appendix D: Description of the manure model 211

Appendix E: The mean values and uncertainty ranges of the LCA model parameters for Eastern Canada 222

Appendix F: Price ratios used for diet formulation (Chapter 4) 232

Appendix G: Co-product allocation data 233

Appendix H: Regional price ratios used for diet formulation (Chapter 5) 234

Appendix I: Ingredient inclusion limits 236

List of Figures

Figure 1.1 The fundamental stages of an LCA according to ISO 14040 (International Organisation for Standardisation 2006a)..3

Figure 2.1 Flow diagram of the boron production system adapted from (Azapagica & Clift 1999a)...17

Figure 2.2 A simplified schematic of energy flow and other causal relationships in animals in livestock production systems. Inputs to the system are indicated in italics while potential co-products are in red bold font...19

Figure 2.3 A simplified schematic of energy flow and other causal relationships in crop production. Inputs to the system are indicated in italics while potential co-products are in red bold font...20

Figure 3.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment Model...31

Figure 3.2 A Monte Carlo comparison of the environmental impacts (Y axis) modelled for Eastern and Western Canadian pig farming systems. The figure shows the % of instances in 1000 Monte Carlo simulations of both systems where impact of either Eastern or Western systems was greater. Any parameter which was a shared input between the two systems returned an identical value from its distribution for each individual comparison. Thus each comparison is based only on the differences between the systems. The distributions for manure application emission factors were considered as uncertainties which were shared between both systems in this case. 41

Figure 3.3 A Monte Carlo comparison of the environmental impacts (Y axis) modelled for Eastern and Western Canadian pig farming systems. The figure shows the % of instances in 1000 Monte Carlo simulations of both systems where impact of either Eastern or Western systems was greater. Any parameter which was a shared input between the two systems returned an identical value from its distribution for each individual comparison. Thus each comparison is based only on the differences between the systems. The distributions for manure application emission factors were **not** considered as uncertainties which were shared between both systems in this case. 42

Figure 5.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment Model. Feed production in the model included the manufacture of fertilisers and pesticides etc. as inputs to growing crops. 83

Figure 5.2 Schematic of the methodology followed in this study to formulate diets for environmental impact objectives. 92

Figure 5.3 The environmental impacts, feed cost and feed intake per kg of Carcass Weight for grower/finisher diets in Eastern Canada formulated for different objectives, represented as a fraction of the results for the least cost diet. Least cost = least feed cost per kg live weight gain, Least cost EFF = least cost / kg Live weight gain while maximising feed efficiency within commercial constraints, NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. Least EI = least combined environmental impact score. 102

Figure 5.4 The environmental impacts, feed cost and feed intake per kg of Carcass Weight for grower/finisher diets in Western Canada formulated for different objectives, represented as a fraction of the results for the least cost diet. Least cost = least feed cost per kg live weight gain, Least cost EFF = least cost / kg Live weight gain while maximising feed efficiency within commercial constraints, NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. Least EI = least combined environmental impact score. 105

Figure 6.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment model. 117

Figure 6.2 Schematic of the methodology followed to formulate diets in different environmental tax scenarios and then test these diets in the life cycle assessment model to determine the resulting environmental impacts. 122

Figure 6.3 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent. C$ = Canadian Dollars 127

Figure 6.4 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq). NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. C$ = Canadian Dollars 128

Figure 6.5 The relative reduction in GWP per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of carbon tax compared to a no tax in scenarios modelled for Eastern Canada and Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq). C$ = Canadian Dollars 129

Figure 6.6 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field in manure. C$ = Canadian Dollars. 131

Figure 6.7 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field in manure NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. C$ = Canadian Dollars.132

Figure 6.8 The relative reduction in N excretion per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of N tax compared to no tax for scenarios in Eastern Canada and Western Canada. N tax levels are shown in C$ per kg of N in manure spread to field. C$ = Canadian Dollars. 134

Figure 6.9 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of phosphorus tax in a) Eastern Canada; b) Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in manure. 136

Figure 6.10 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of phosphorus tax in a) Eastern Canada; b) Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in manure NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential.. 137

Figure 6.11 The relative reduction in P excretion per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of P tax compared to no tax for scenarios in Eastern Canada and Western Canada. P tax levels are shown in C$ per kg of P in manure spread to field. C$ = Canadian Dollars. 138

List of Tables

Table 1.1. Summary of the allocation approaches adopted in Life Cycle Assessment (LCA) studies of pig farming systems. The allocation methodology adopted in two model areas where allocation is most commonly applied in LCA of pig farming systems: feed supply chain and manure application is shown in each case. (adapted from a table produced for FAO 2016b) 8

Table 3.1 The mean, maximum (max) and minimum (min) values fitted from the benchmark data[1] for herd performance characteristics in Eastern and Western systems. 35

Table 3.2 The environmental impacts of Eastern Canadian pig farming systems as outcomes of LCA; impacts are separated by source category and production stage. The mean, upper and lower values for each impact category at 95% confidence levels are shown with both α and β uncertainties considered. 43

Table 3.3 The environmental impacts of Western Canadian pig farming systems as outcomes of LCA; impacts are separated by source category and productions stage. The mean, upper and lower values for each impact category at 95% confidence levels are shown with both α and β uncertainties considered. 45

Table 3.4 A sensitivity analysis of the parameters in the LCA, based on the typical pig production system for Eastern Canada. Parameters were tested to the upper and lower 95% confidence intervals of their distributions (except in the case of triangular distributions) in order to assess the largest sources of uncertainty in the LCA. Only cases where parameter variability caused result to move +/- 5% from mean are shown. 48

Table 4.1 The overall ingredient and nutritional composition (across all 4 feeding phases) of the grower/finisher diets tested in Experiment 1. The meat meal, bakery meal, corn DDGS and wheat shorts diets were the outcome of least cost formulations which included the maximum amount of these co-products. All ingredient inclusions shown in g/kg as fed; all nutrient levels shown as % as fed unless otherwise stated. 61

Table 4.2 The overall ingredient and nutritional composition (across all 4 feeding phases) of the grower/finisher diets tested in Experiment 2. The OP diet was a least cost formulation designed for Optimum Feed Efficiency. The subsequent diets shown were formulated at 97.5%, 95% and 92.5% the nutritional density of the OP diet (the 0.975 OP. 0.95 OP and 0.925 Op diets). All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated. 63

Table 4.3 Maximum inclusion limits (g/kg as fed) used in each feeding phase in the grower finisher diets for the co-products investigated in this study 64

Table 4.4 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets in the scenarios tested. 67

Table 4.5 The average levels of environmental impact **per kg of feed** for grower/finisher diets tested Canadian pig production. The meat meal, bakery meal, corn DDGS and wheat shorts diets were least cost formulations which included the maximum amount of these co-products 71

Table 4.6 The environmental impacts of **1 kg expected carcass weight** at farm gate for grower/finisher control and co-product diets tested in an LCA of Canadian pig production.. The meat meal, bakery meal, corn DDGS and wheat shorts diets were least cost formulations which included the maximum amount of these co-products. The control diet was a simple corn based diet containing none of these ingredients 72

Table 4.7 The environmental impact of 1 kg **expected carcass weight** at farm gate for grower/finisher diets Canadian pig production. The OP diet was a least cost formulation designed for Optimum Feed Efficiency. The other three diets shown were formulated at 97.5%, 95% and 92.5% the energy density of the OP diet (the 0.975 OP. 0.95 OP and 0.925 OP diets).This allow for a higher inclusion of co-products in these diets...............................74

Table 5.1 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets tested. Inventory data for these ingredients was compiled as part of previous life cycle assessment studies of Canadian pig farming systems (see Chapter 3 & 4)85

Table 5.2 The nutritional specifications of the “typical” grower/finisher diet for Canadian pig systems. The lower limits permitted in the diet formulation rules used in this study are also shown...91

Table 5.3 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated for different objectives for Eastern Canada. All ingredient inclusion and nutrient levels shown are g/kg as fed unless otherwise stated. The average predicted feed intake and feed costs for each grower/finisher diet are also shown.................96

Table 5.4 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated for different objectives for Western Canada. All ingredient inclusion and nutrient levels shown are g/kg as fed unless otherwise stated. The average predicted feed intake and feed costs for each grower/finisher diet are also shown.................98

Table 5.5 The environmental impacts per kg of Carcass Weight for grower/finisher diets in Eastern Canada formulated for different objectives. 101

Table 5.6 The environmental impacts per kg of Carcass Weight for grower/finisher diets in Western Canada formulated for different objectives. 104

Table 6.1 The nutritional specifications of the "typical" grower/finisher diet for Canadian pig systems. The lower limits permitted in the diet formulation rules used in this study to ensure feed intake was not affected by issues such as gut fill are also shown. 118

Table 6.2 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets tested. Inventory data for these ingredients was compiled as part of a previous life cycle assessment studies of Canadian pig farming systems (see Chapters 3 & 4). 124

Chapter 1: Introduction

1.1 The environmental impact of livestock systems

The global livestock sector is undergoing a period of increased demand and rapid expansion; by 2050 it is predicted to have reached an annual output of 465 million tonnes of meat, more than double the 229 million tonnes produced in 2000 (Steinfeld et al., 2006). The environmental impact of livestock farming systems, combined with the increasing human demand for livestock products has become an important aspect of the current debate on food security and the future of the human food supply chain (Herrero et al., 2013; Springmann et al., 2016; Steinfeld et al., 2006). In general, livestock production is known to use a large proportion of globally available freshwater (≈20%) and ice free land (≈30%), have important implications for biodiversity, and contribute around 15-20% of global greenhouse gas (GHG) emissions (Mekonnen and Hoekstra, 2012; Springmann et al., 2016; Steinfeld et al., 2006; Vermeulen et al., 2012). Livestock can provide essential nutrients to crop production, but can also be a major source of pollution, emitting organic matter, minerals and pathogens into rivers, lakes and coastal areas (Herrero et al., 2013). Greater scrutiny has been placed on the livestock industry regarding its environmental impact in the last decade since the release of the FAO report entitled "Livestock's long shadow" (Steinfeld et al., 2006). This has led to an increased interest in the use of quantitative models to assess the environmental impact of livestock production and (potentially) identify appropriate technical adjustments to reduce it. Modelling environmental impacts can involve a large array of approaches, from simply quantifying the emissions of a particular compound at one point during the production cycle, to more complex approaches such as modelling eco-system services or life cycle assessment covering the whole production chain.

1.2 The environmental impacts of pig farming systems

Livestock production systems and their resulting impact on the local environmental are extremely diverse both between and within species. The high profile nature of climate change issues has meant there has been more focus on modelling the environmental impact of beef and dairy production systems, because ruminants account for the majority of GHG emissions caused by livestock systems (Ripple et al., 2014; Steinfeld et al., 2006). However, pork is the world's most widely consumed source of animal protein representing 37% of the meat produced globally in 2011 (Macleod et al., 2013). Pig farming systems are associated with other important environmental concerns; these include resource inputs to the animal feed supply chain and the contribution of nitrogen (N) and phosphorus (P) excreted in pigs manure to eutrophication when spread as fertilizer (Basset-Mens and Van Der Werf, 2005; Eshel et al., 2014; Macleod et al., 2013; Thoma et al., 2013). Pig farming systems are associated with some of the highest levels of acidification and eutrophication of any species in the livestock sector (de Vries and de Boer, 2010) The production of feed is generally considered to be the largest contributor to the environmental impact of pig farming systems (Basset-Mens and Van Der Werf, 2005; Nguyen et al., 2011). The pig feed supply chain can involve large levels of resource input as pig diets are usually based on cereals grown for animal feed which require fertilizers, pesticides and other resources as part of their production. Emissions which occur inside pig housing, both while manure is stored and when it is applied to field as fertilizer are also very important when quantifying the eutrophication and acidification caused by pig farming systems (McAuliffe et al., 2016). The levels of environmental impact which result from manure management are sensitive to the storage and treatment technologies used (Prapaspongsa et al., 2010). However, levels of nutrient excretion by pigs in manure are greatly affected by the amount of nutrients fed in pig diets, thus feeding decision also has implications for environmental impacts caused by manure management. As such the ingredient and nutritional composition of the pig diets are extremely important considerations when quantifying the environmental impacts of pig farming.

1.2.1 Canadian pig farming systems

Globally Canada has the 10th largest pig farming sector by country and is the 5th largest exporter of products from pig production (FAOSTAT, 2016). Pig production practices in Canada are similar to other large producers in the developed world such as Germany and the United States, with large herds and a heavy focus on improving production efficiency and optimising profitability (Brisson, 2014). In Canada, the pig sector has faced scrutiny regarding its contribution to the oversupply of nitrates and phosphorus to freshwater systems resulting from the spreading of manure on cropped fields (Pomar et al., 2007). There have been very few peer reviewed studies which quantified the environmental impact of Canadian pig farming systems, with only Vergé et al. (2009) publishing an estimation of GHG emissions caused by the Canadian pig sector. Pig diets in Eastern Canada are typically based on corn similar to USA pig diets (Thoma et al., 2011), whereas pig diets in Western Canada use wheat and barley as the main cereal components (Patience et al., 1995), as would be common for European pig diets. The contrasting typical feed ingredients used in pig systems in Eastern and Western Canada make it an interesting system to model and compare the potential of dietary change to reduce environmental impacts in these two regions.

1.3 Life Cycle Assessment

Life Cycle Assessment (LCA) is a generally accepted method to evaluate holistically the environmental impact during the entire life cycle of a product or system (Guinée et al., 2002). A LCA is defined as "the systematic evaluation of the environmental aspects of a product or service system through all stages of its life cycle." (United Nations Environment Programme, 2016). Figure 1.1 represents the traditional framework which LCA practitioners follow in accordance with the international standard on principles and framework for LCA modelling - ISO 14040 (International Organisation for Standardisation, 2006a).

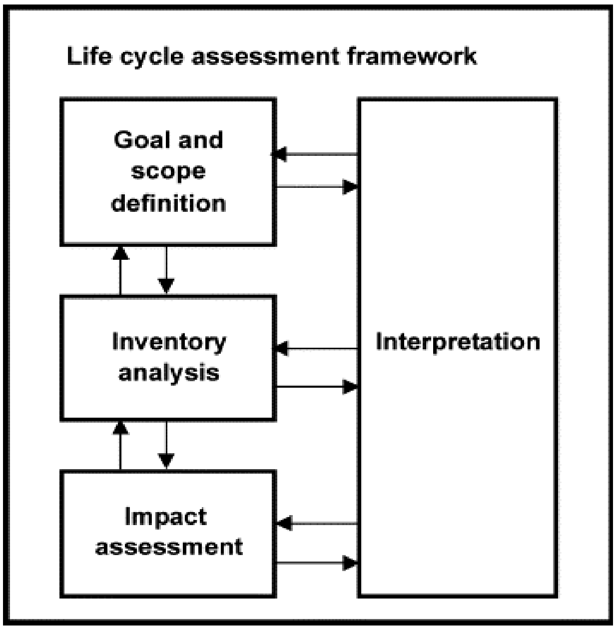


Figure 1.1 The fundamental stages of an LCA according to ISO 14040 (International Organisation for Standardisation, 2006a).

LCA modelling has four phases under this framework; Goal and scope definition, inventory analysis, impact assessment and interpretation. Almost all major decisions on the design of an LCA should be based on the initially defined goal and scope of the study, this is regarded as the most important stage of the process (Andersson et al., 1994)**.** These decisions involve defining the functional unit upon which impacts will be assessed as well as the system boundary for the LCA model. The next stage, inventory analysis involves collating all relevant data on the system modelled regarding its inputs and outputs, including emissions during a process and waste disposal to establish a Life Cycle Inventory (LCI). Impact assessment is then carried out during which scientifically defined characterisation factors are applied to different emissions and resource inputs to the production system in order to quantify its overall environmental impact. This is then typically reported using a set of impact categories such as Global Warming Potential (GWP) or Eco-toxicity (Williams, 2009). Throughout all of these stages, authors of LCA study are expected to systematically identify, qualify, check and evaluate the methodological choices made at each stage in order to properly interpret the results. This is required under ISO 14044 guidelines on LCA and often includes checking the model for methodological consistency, a sensitivity analysis, an uncertainty analysis and openly presenting the limitations of the methodological approach in any final reporting (International Organisation for Standardisation, 2006a; Williams, 2009).

1.3.1 Life Cycle Assessment of pig production systems

The livestock industry has positively engaged with LCA as an appropriate tool to quantify and reduce the environmental impacts of livestock farming. Many industry levy boards have funded LCA studies to understand the environmental impact of individual sectors of the livestock industry at a country level (e.g. Kingston et al., 2009; Leinonen et al., 2012; Thoma et al., 2011; Wiedemann et al., 2010). At the international level, the livestock environmental assessment and performance partnership (LEAP) has been established since 2010 (FAO, 2016a). LEAP has recently published draft guidelines on modelling the environmental impacts of pig systems using LCA. The guidelines cover four environmental impact categories GWP, eutrophication potential (EP), non-renewable energy use (NRE) and water use which were identified as important for pig farming systems (FAO, 2016b). The impact categories GWP, Acidification Potential (AP) and EP are most commonly used in LCA studies of pig systems (McAuliffe et al., 2016).

LCA studies define their functional unit and system boundaries based on the scope of the study. While some LCAs of pig production focus solely on modelling the environmental impacts that occur in the feed supply chain or manure management systems (McAuliffe et al., 2016) the approach taken in this thesis was to model the environmental impact of pig production systems from cradle to farm-gate. This encompassed activities such as inputs to feed production, feed processing, on farm energy use, manure storage and disposal, to name a few. A functional unit of 1 kg expected carcass weight (ECW) was used in the LCA model throughout the thesis; functional units based on live weight, or carcass weight are commonly used in cradle to farm-gate LCA of pig systems (McAuliffe et al., 2016).

1.2.3 Reducing the environmental impacts of pig systems through dietary change

Several LCA studies have considered the potential for dietary change to reduce the environmental impacts of pig production. Generally these can be classified in two groups: 1) those which tested the effect of increased amino acid supplementation (Garcia-Launay et al., 2014; Mosnier et al., 2011a; Ogino et al., 2013) and 2) those which investigated the use of alternative protein sources to replace soybean meal in European systems (Cederberg et al., 2005; Eriksson et al., 2005; Meul et al., 2012; Reckmann et al., 2016). Due to the pressure of the animal feed supply chain on human food systems (Steinfeld et al., 2006), there is an increased interest in the use of alternative feed ingredients such as by-products from the human food supply chain in livestock diets (Woyengo et al., 2014; Zijlstra and Beltranena, 2013). However, the consequences of including of such co-products in pig diets have so far received very little attention in LCA models of pig farming systems and should be investigated systematically.

LCA models of livestock systems have commonly been used to test the effect of specific dietary changes on the environmental impacts. In some rare cases researchers have gone one step further and integrated LCI data into a diet formulation algorithm in order to formulate diets to explicitly minimise specific environmental impacts in poultry systems (Moe et al., 2014; Nguyen et al., 2012). However, these studies had key methodological limitations; firstly, only the environmental impact per kg of feed was minimised, the implications of the diets formulated for nutrient excretion were not considered. Effectively, the fate of the feed once fed to the animal in its effect on animal performance and predicted nutrient excretion was outside the system boundary of these studies. This could mean that diets that minimised specific environmental impacts in feed production actually increased the environmental impacts caused by emissions during manure management, which contribute a large proportion of the acidification and eutrophication in pig and poultry production systems (de Vries and de Boer, 2010). The effects of any potential diet on nutrient excretion and emissions must be considered when looking to minimise the environmental impact of livestock systems using diet formulation.

Secondly, in these studies diets were formulated for a fixed minimum nutritional specification for energy (MJ/kg) and nutrient content (g/kg) above which feed intake was assumed to be unaffected. This is a fairly restrictive way to formulate diets, and further does not consider the trade-off between environmental impact per kg of feed and feed intake. When formulating pig diets in a commercial setting, the diets are formulated for economic objectives and in many cases the trade-off between feed cost and feed efficiency is considered as part of this (Ferguson, 2014). Using this approach, optimum nutrient to energy ratios can be defined, the expected effect of energy density on feed intake can be accounted for, and a range of energy densities can be permitted when formulating animal diets. This allows a feed formulation algorithm to identify the "optimum" energy density for a diet under different circumstances (for e.g. as prices fluctuate) or for different objectives (Ferguson, 2014). Livestock diets have not been previously formulated for environmental impact objectives using this flexible approach to the nutritional density of the solution. The trade-off between nutritional specification (and resulting feed intake) and the environmental impact per kg of livestock diets is another area which requires further investigation.

A recent study, which formulated pig diets to reduce GWP, was able to account for the two methodological challenges described above by utilising a previously developed animal growth model as part of a diet formulation tool (Garcia-launay et al., 2015). However, one methodological challenge which had still not been addressed in this area is how to formulate livestock diets for environmental impact objectives which consider more than one impact category. The trade-offs when reducing different types of environmental impact through diet optimisation in livestock systems have needed to be explored.

1.4 Methodological challenges for livestock LCA modelling

Models of agricultural systems increasingly account for a large proportion of the total peer reviewed LCA studies; since 2007 around 25% of studies published in the International Journal of LCA are related to agriculture, about double the proportion before 2007 (Durlinger et al., 2014). However, agricultural systems are often complex and this can present issues when trying to model their environmental impacts in a LCA. The increasing number of agricultural studies in recent years has highlighted some important methodological challenges relating to LCA modelling of these systems. Two of these, explored in further detail in this thesis, are co-product allocation and uncertainty analysis.

1.4.1 Co-product allocation

Co-product allocation is defined as "objectively assigning resource use, energy consumption, and emissions to identified co-products where there is no physical or chemical way to separate the activities that produce them" (USDA, 2014). This is necessary in instances where a process being modelled has multiple outputs which are considered to have a function or value. Co-product allocation is a key concept within Life Cycle Assessment (LCA) (Suh et al., 2010) and is one of the most discussed methodological issues in the field (Finnveden et al., 2009; Guinée et al., 2011). Agricultural systems are complex and involve many multi-functional processes making allocation a key aspect of LCA modelling in agriculture. Studies of various agricultural systems and sectors have shown the sensitivity of results to methodological choices made regarding co-product allocation (Flysjö et al., 2011; Nguyen et al., 2011; Van Der Werf and Nguyen, 2015; Wiedemann et al., 2015) As such it is important that LCA models of agricultural systems adopt allocation methodologies which are appropriate for the system being modelled and transparent. For LCA of livestock systems, the varied methodologies adopted for co-product allocation have been highlighted as one of the main reasons for the high levels of variability seen in LCA results of specific livestock supply chains (Notarnicola et al., 2015).

Table 1.1 summarises the different approaches adopted in recent LCA studies of pig systems to deal with the issue of co-product allocation. The methodologies adopted can be broadly classed into three groups: 1) avoiding co-product allocation through adopting system expansion or system separation, 2) allocation based on shared physical properties such as mass or gross energy, and 3) allocation between co-products based on the price they command (commonly known as "economic allocation"). As can be seen in Table 1.1, economic allocation is the most commonly used methodology for allocation for the feed supply chain in pig LCA studies. Economic allocation is also recommended as the most appropriate methodology for allocation for the feed supply chain by the LEAP partnership (FAO, 2014a). However, some researchers have suggested that using economic allocation is undesirable in LCA (Ayer et al., 2007; Pelletier and Tyedmers, 2011) and many new methodologies for co-product allocation in livestock LCA based on physical properties or relationships are being developed (Eady et al., 2012; Gac et al., 2014; International Dairy Federation, 2010; Thoma et al., 2013; Van Der Werf and Nguyen, 2015; Wiedemann et al., 2015). This methodological trend warranted examination, as the methodology adopted in this sensitive area of LCA modelling has implications for all the results produced in this thesis.

Table 1.1 Summary of the allocation approaches adopted in Life Cycle Assessment (LCA) studies of pig farming systems. The allocation methodology adopted in two model areas where allocation is most commonly applied in LCA of pig farming systems: feed supply chain and manure application is shown in each case. (adapted from a table produced for FAO 2016b)

Study	Functional Unit	Allocation system - feed supply chain	Allocation system - manure application
(Cederberg and Flysjö, 2004)	1kg meat - fat + bone free	Economic[1]	System separation (to crops)
(Basset-Mens and Van Der Werf, 2005)	1 kg Live Weight (LW)	Economic	System separation (to crops)
(Eriksson et al., 2005)	1 kg LW	Economic	System Separation (to crops)
(Williams et al. 2006)	1 tonne Carcass Weight (CW)	Economic	System Expansion (credits for reducing fertilizer application)
(Dalgaard et al., 2007)	1 kg CW	System Expansion	System Expansion
(Cederberg et al., 2009)	1 kg CW	Economic	System separation (to animal production)
(Kool et al., 2009)	1 kg CW	Economic	Physical property (based on active N content)
(Olea et al., 2009)	1 tonne LW	System Expansion	System Expansion
(Halberg et al., 2010)	1 kg LW	System Expansion	System Expansion
(Pelletier et al., 2010)	1 kg LW	Physical property (Gross Energy)	System Separation (to animal production)

Study	Functional Unit	Allocation system - feed supply chain	Allocation system - manure application
(Stone et al., 2010)	89 kg LW	Physical property (Mass)	System Expansion (credits for reducing fertilizer application)
(Wiedemann et al., 2010)	1 tonne CW	System expansion[1,2]	System expansion
(Nguyen et al., 2011)	1 kg CW	System Expansion[2]	System expansion
(Thoma et al., 2011)	4 oz Boneless Pork [3]	Economic	System Separation (to animal production)
(Weiss and Leip, 2012)	1 kg CW	Physical property (N content)	System expansion
(Ogino et al., 2013)	115 kg LW	Economic	Economic
(Reckmann et al., 2013)	1 kg CW	Economic	System expansion
(Cherubini et al., 2015)	1 tonne CW	Economic	System Expansion

1. Comparison with an attributional approach based mass allocation included
2. Some instances of mass allocation included within the consequential framework in the feed supply chain
3. The results from cradle to farm-gate for the function unit 1kg LW also included

1.4.2 Modelling uncertainty

Accounting for uncertainty in LCA is important to produce credible and reliable results (Lloyd and Ries, 2007). Input data to LCA models is often highly variable, particularly for agricultural systems; results presented as single point values which overlook this are misleading (Groen et al., 2014a). LCA is commonly used as a decision support tool and, as in any quantitative research, statistical comparison of alternative scenarios is only possible if the uncertainty range of the results are calculated (Leinonen et al., 2013). Guidelines from the FAO LEAP committees for LCA of multiple livestock sectors now recommend that "wherever data is gathered, data should also be collected for uncertainty assessment" (FAO, 2014a, 2014b, 2014c). However, despite this, systematic, quantitative uncertainty analyses have rarely been applied in LCA studies of agricultural systems (Leinonen et al., 2013). LCA of agricultural systems can be complex with thousands of underlying unit processes being shared between any systems which are compared. This complexity acts as barrier to modelling uncertainty in LCA; it can make it necessary to estimate the variability of a large number of parameters, and make the computational requirements for repeat simulations of LCA models prohibitive (Groen et al., 2014a). It is important that techniques for uncertainty analysis adopted in LCA studies of agricultural systems can overcome these difficulties to identify where real differences exist between the systems being compared.

Recently, a methodology for uncertainty analysis designed to address this issue using Monte-Carlo simulations was implemented in a LCA of UK poultry systems (Leinonen et al., 2012). This method separated uncertainties within the model that were specific to one scenario modelled (α uncertainty), and those which were shared between two or more scenarios which were being compared (β uncertainty). While this was a significant step forward in terms of modelling uncertainty in LCA of agricultural systems, it is not without its disadvantages. For example, it didn't include shared uncertainty in comparisons between two production scenarios, thus assuming their calculated environmental impacts were affected evenly by this uncertainty. However, this may not be the case, for example uncertainty in the predicted yield of a crop included in an animal diet for two production scenarios is shared between these systems, but will cause different levels of uncertainty in the calculation of their impacts if the crop makes up a larger proportion of the diet in one of the scenarios than the other. Further development of methodologies to address this important issue is required for agricultural LCA studies to be used effectively as decision support tools.

1.5 Thesis aims

The primary aim of this thesis was to model the environmental impacts of pig farming systems in Canada using LCA and to quantify the potential for nutritional solutions to reduce the environmental impact of the system. The thesis also aimed to address some of the important methodological issues regarding the LCA modelling of livestock systems, as these would affect the reported outcomes of any LCA study on this subject. The specific aims of the thesis Chapters were:

1) To review the latest methodologies being proposed for co-product allocation in livestock LCA studies and assess which was the best approach to adopt in this project (Chapter 2).

2) To use industry benchmark data to develop an LCA model to quantify the environmental impacts of typical pig production systems in Canada for multiple environmental impact metrics. As well as this, to develop a methodology for uncertainty analysis which could provide meaningful comparisons between two pig farming systems while accounting for the high levels of variability in these systems (Chapter 3).

3) To investigate the effect of including specific co-products in grower/finisher (G/F) diets, and the effect of reducing the energy density (and therefore the feed efficiency of the animals), whilst increasing co-product levels of G/F diets on the environmental impacts of pig systems (Chapter 4).

4) Develop a methodology which enables pig diets to be formulated explicitly for environmental impact objectives using an LCA approach, including the environmental impacts caused by nutrient excretion for different diet scenarios. (Chapter 5).

5) To quantify the potential effect of environmental taxes on the composition of pig diets, and the implications for the environmental impacts of the production system if such taxes were accounted for directly in a diet formulation algorithm (Chapter 6).

Chapter 2: The need for co-product allocation in the Life Cycle Assessment of agricultural systems – is "biophysical" allocation progress?

2.1 Abstract

Several new "biophysical" co-product allocation methodologies have been developed for LCA studies of agricultural systems based on proposed physical or causal relationships between inputs and outputs (i.e. co-products). These methodologies are thus meant to be preferable to established allocation methodologies such as economic allocation under the ISO 14044 standard. The aim here was to examine whether these methodologies really represent underlying physical relationships between the material and energy flows and the co-products in such systems, and hence are of value. Two key components of agricultural LCAs which involve co-product allocation, were used to provide examples of the methodological challenges which arise from adopting biophysical allocation in agricultural LCA: 1) The crop production chain and 2) The multiple co-products produced by animals. The actual "causal" relationships in these two systems were illustrated, the energy flows within them detailed and the existing "biophysical" allocation methods, as found in literature, were critically evaluated in the context of such relationships. The premise of many biophysical allocation methodologies has been to define relationships which describe how the energy input to agricultural systems is partitioned between co-products. However, we described why none of the functional outputs from animal or crop production can be considered independently from the rest on the basis of the inputs to the system. Using the example of manure in livestock systems, we also showed why biophysical allocation methodologies are still sensitive to whether a system output has economic value or not. This sensitivity is a longstanding criticism of economic allocation which is not resolved by adopting a biophysical approach. The biophysical allocation methodologies for various aspects of agricultural systems proposed to date have not adequately explained how the physical parameters chosen in each case represent causal physical mechanisms in these systems. Allocation methodologies which are based on shared (but not causal) physical properties between co-products are not preferable to allocation based on non-physical properties within the ISO hierarchy on allocation methodologies, and should not be presented as such.

2.2 Introduction

Co-product allocation is defined in the ISO series of international standards on LCA as "partitioning the input or output flows of a process or a product system between the product system under study and one or more other product systems" (International Organisation for Standardisation, 2006a, 2006b). Originating from practices in economics and other management sciences, co-product allocation is a key concept within Life Cycle Assessment (LCA) (Frischknecht, 2000; Suh et al., 2010) and is one of the most discussed methodological issues in the field (Finnveden et al., 2009; Guinée et al., 2011; Hanes et al., 2015; Heijungs and Frischknecht, 1998). Recently there has been a considerable effort by researchers and industry funded committees (such as the FAO Livestock Environmental Assessment and Performance partnership (LEAP)) to establish the most appropriate allocation methodology for LCA studies of livestock production (FAO, 2014a, 2014b, 2014c; International Dairy Federation, 2010). This has been part of wider efforts to unify methodologies adopted by those developing LCA models of agricultural and in particular livestock systems to ensure they are comparable in their approach since it is obvious that the use of different allocation rules in LCA studies comparing different aspects of agricultural systems can led to different conclusions (Brankatschk and Finkbeiner, 2014; Eady et al., 2012; Nguyen et al., 2011).

The allocation of environmental impacts to co-products based on their economic value is the most commonly used allocation method in agricultural LCA studies, particularly for crop production and the livestock feed supply chain (Ardente and Cellura, 2012; Brankatschk and Finkbeiner, 2014; Van Der Werf and Nguyen, 2015). However, several new allocation methodologies have been proposed for LCA studies of agricultural systems based on physical relationships between co-products. These methodologies are often referred to as "biophysical" allocation (Eady et al., 2012; Gac et al., 2014; International Dairy Federation, 2010; Thoma et al., 2013; Van Der Werf and Nguyen, 2015; Wiedemann et al., 2015). Draft guidelines on carbon footprinting in livestock systems issued by the FAO have also recommended that biophysical allocation should be adopted for models of the on-farm stages of livestock production (FAO, 2014b, 2014c), although currently not in the feed supply chain (FAO, 2014a). These developments have followed from the ISO standard of requirements and guidelines for LCA, which state that co-product allocation based on underlying physical relationships between the material flows of a system and its products or functions is preferable to allocation based on other relationships, such as economic value (International Organisation for Standardisation, 2006b).

The methodological trend towards biophysical allocation in agricultural LCA raises obvious and wider questions: what can be considered an underlying physical relationship between material flows and productive outputs in LCA? Are such relationships easily related to the outputs of agricultural systems which are useful from a human perspective? Ultimately, is “biophysical” allocation an appropriate approach for LCA of agricultural systems? The aims of this paper were to 1) examine whether researchers have been able to identify underlying physical relationships between the material and energy flows of agricultural systems and their products and 2) assess whether the trend towards biophysical allocation in agricultural LCA is feasible from a methodological perspective. Two key components of agricultural systems which involve co-product allocation were used to provide examples of current methodological practices and issues namely: 1) The crop production chain and 2) The multiple co-products produced by livestock.

2.3 Co-product allocation and its use in agricultural LCA

ISO 14044 is the international standard of requirements and guidelines for best practice in conducting LCA (International Organisation for Standardisation, 2006b). Part of the standard sets out a hierarchy for the methodological choices available regarding co product allocation in LCA:

a) **Step 1**: Wherever possible, allocation should be avoided by

1) dividing the unit process to be allocated into two or more sub-processes and collecting the input and output data related to these sub-processes, or

2) expanding the product system to include the additional functions related to the co-products, taking into account the requirements of the ISO guidelines on system boundaries.

b) **Step 2**: Where allocation cannot be avoided, the inputs and outputs of the system should be partitioned between its different products or functions in a way that reflects the *underlying physical relationships* between them; i.e. they should reflect the way in which the inputs and outputs are changed by quantitative changes in the products or functions delivered by the system.

c) **Step 3**: Where physical relationship alone cannot be established or used as the basis for allocation, the inputs should be allocated between the products and functions in a way that reflects other relationships between them. For example, input and output data might be allocated between co-products in proportion to the economic value of the products. (International Organisation for Standardisation, 2006b).

The ISO standard suggests that co-product allocation is to be avoided wherever possible in LCA decision making. However the adoption of either system separation or system expansion throughout entire LCA models can require large amounts of extra data to model either additional sub-processes or marginal systems (Curran, 2015; Parker, 2008), as most processes modelled in LCA are multi-output (Frischknecht, 1994). Aside from the practical issue of obtaining extensive datasets, these large complex models also run the risk of being less transparent and using inaccurate assumptions (Curran, 2007; Ekvall, 1999). While system expansion is generally associated with consequential LCA modelling, it is also utilised in many attributional models (Finnveden et al., 2009). For example, many attributional livestock LCA have used system expansion to account for nutrients in manure replacing the need for inorganic fertilizers when spread on fields for crop production (Williams et al. 2006; Reckmann et al. 2013; Cherubini et al. 2015). There are however, wider concerns as to whether implementing system expansion throughout LCA models to avoid co-product allocation is feasible or desirable (Finnveden et al., 2009). In theory, multifunctional processes could be added to LCA studies ad infinitum in order to fully implement this methodology for every aspect of an LCA (Lundie et al. 2007). Unlike the example of manure replacing inorganic fertilizer, there are many areas of LCA models of livestock systems where such "what if" exercises are purely speculative. For example when utilising co-products such as corn or wheat dried distillers grains with solubles from bioethanol production in animal feed there are a multitude of pathways for such material to be used if not included in the diets for the particular livestock system modelled. Expanding the model with a "what if" scenario to predict the replacement pathway for a particular ingredient when this cannot be predicted with any confidence means the modelling exercise strays further away from using known facts (Heijungs and Guinée, 2007). With this in mind it is not possible or desirable to use system expansion as a general rule to eliminate allocation problems throughout LCA models of agricultural systems

In cases where co-product allocation is necessary, step 2 of the hierarchy advises that inputs and outputs to a system are partitioned in a way that reflects the underlying (or causal) relationships beneath them (Azapagica and Clift, 1999a; Ekvall and Finnveden, 2001). This recommendation of the ISO standard is a significant reason why many of the new "biophysical" methodologies for allocation discussed below have been proposed as preferable to allocation based on the economic value of co-products. In this sense the hierarchy followed the recommendations of a number of papers on the subject which used industrial processes (Azapagica and Clift, 1994; Clift et al., 1996) and was the outcome of recommendations made by the working groups of bodies, such as the Society of Environmental Toxicology And Chemistry (SETAC) (SETAC, 1994). Similar recommendations regarding a hierarchy for allocation methodologies in agricultural LCA can be found in reports from an EU working group on methodology harmonisation (Audsley et al., 1997).

2.4 Allocation using underlying physical relationships in an industrial setting

Many of the conventional guidelines and practices within the field of LCA should be viewed in the context of its early history, during which it was mainly used as a tool to measure energy use and resource consumption from large industrial processes (European Environment Agency, 1999). In the case of co-product allocation, it is very plausible that in most cases allocation can be avoided through system separation in an LCA of a large industrial production process (Azapagica and Clift, 1999b). Large industrial production sites typically have large amounts of instrumentation and data on the exact inputs and outputs from production processes. The causal mechanisms behind these production processes are in many cases well known and can be defined by process engineers. Example 1, originally presented by Azapagica and Clift (1999a) briefly describes an allocation methodology based on causal physical relationships approach being applied to a mineral processing facility producing five boron co-products using linear programming to model system behaviour.

2.4.1 Example 1 – Allocation in the boron co-product system (Azapagica and Clift, 1999a)

The boron production system shown in Figure 2.1 has 5 boron co-products:1) Disodium terraborate decahydrate ($Na_2B_4O_710H_2O$) "10 Mol", 2) Disodium terraborate pentahydrate ($Na_2B_4O_74.67H_2O$) "5 Mol", 3) Boric acid (H_3BO_3) "BA", 4) Anhyrdrous Borax ($Na_2B_4O_7$) "AB", and 5) Anhyrdrous Boric acid (B_2O_3) "ABA". As shown in Figure 2.1, the LCA was split into a "foreground system", which was the boron mine and production plant and a "background system", which comprised all other activities from material extraction to delivery to the foreground system. In the foreground system, the minerals borax and kernite are extracted from the mine, crushed and transported to an adjacent plant. 10 and 5 Mol borates are produced by dissolving borax and kernite in water. BA is produced separately by reacting kernite ore with sulphuric acid and AB and ABA are produced in high temperature furnaces from 5 Mol borate and BA respectively. All products are then shipped from the factory gate. Electricity and steam for the system are provided by the on-site natural gas co-generation plant. All activities except the disposal phases of these products are considered in this cradle to gate LCA (Azapagica and Clift, 1999a).

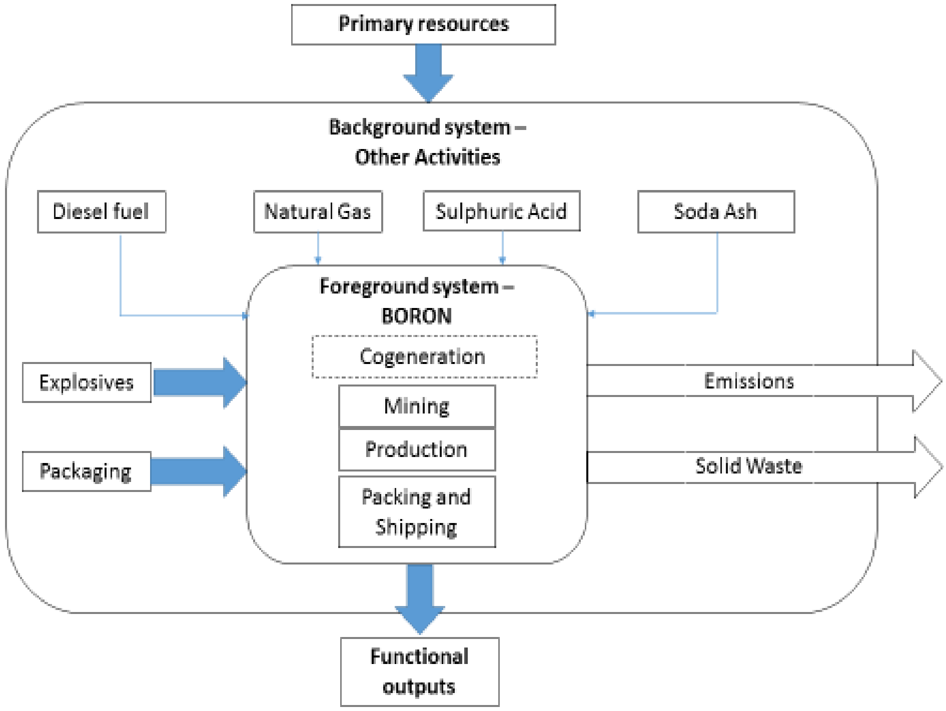


Figure 2.1 Flow diagram of the boron production system adapted from (Azapagica and Clift, 1999a).

Azapagica and Clift used linear programming (LP) to model physical relationships in the boron system so that infinitesimal variations in the functional outputs were modelled to determine "marginal allocation coefficients". The relationships which described the system behaviour were modelled using constraints in the LP algorithms. Upon providing a solution, the LP model also showed marginal values indicating the contribution of each constraint to the total burdens. Where a constraint limited the behaviour of the system, it had a marginal value greater than 0; non-active constraints had marginal values of 0 and were thus modelled as not contributing to the burdens resulting from the system. In this case the authors assigned environmental impacts to constraints related to co-product outputs which are considered to be active and thus contributing to the environmental burdens. Any limits imposed by aspects of the production process were ignored. The marginal approach allowed the model to allocate the environmental impacts on the basis of the expected increase in emissions or resource input required to produce additional yield of each co-product. In this system most of the CO_2 emissions were allocated between AB and ABA as increasing production of either of these co-products requires large energy inputs to a furnace as well as further production of 5 Mol and BA respectively. Further analysis by the authors using alternative co-product allocation methodologies showed that co-product allocation on the basis of mass flow produced the same results as those from marginal allocation using the LP model. As such, the authors were able to demonstrate that allocation using this simple physical property was appropriate to represent the causal mechanisms at work in the production system. However they were only able to select the appropriate property using a holistic model of system behaviour, rather than selecting arbitrarily.

2.5 "Biophysical" allocation in Agricultural LCA

While "causal physical relationships" between the material inputs and the outputs in Life Cycle Assessment have been modelled in industrial processes, the question is whether such an approach can be easily related in the biological systems, which underpin agricultural production. At the organism level, the biological systems do not function with the goal of producing the items which humans deem to be economically valuable (and consider as co-products of the system). In order to establish physical causality between functional units and environmental burdens, it must be possible to change the functional outputs of the product system independently (Azapagica and Clift, 1999b; Ekvall and Finnveden, 2001). In the following, the functioning and relationships of different sub-processes in both crop and animal production are demonstrated and discussed with in the context of physical causalities.

Figure 2.2 is a simplified representation of the energy flow and other causal relationships in animals in livestock production systems. It can be seen in the figure that biological processes involved in animal production form a complex network of interactions and that none of the functional outputs can be considered independently from other outputs or the inputs to the system. All the energy directly utilized by the animals in the production process enters the systems in the form of chemical energy obtained from the feed. This energy can then be considered to be partitioned to different outputs, some of which can be seen as useful, i.e. economically valuable products such as meat (containing proteins and lipids), products obtained when the animal is alive (e.g. eggs and milk), manure (used as fertilizer or as fuel) and animal by-products (i.e. parts of the slaughtered animal not used for human consumption). Other outputs can be unwanted and considered as "waste" and include methane (from enteric fermentation) and energy as heat from metabolic processes. These unwanted outputs cannot be ignored when exploring the "causal" relationships between the biological processes of animal production. For example, the metabolic heat production can be seen as "construction cost" without which the production of useful animal products would not be possible.

Furthermore, it should be also noted that feed (as a source of energy) is not the only input that is directly involved in animal production, especially when the LCA modelling framework is considered. Growing the animal especially in indoor conditions requires a considerable amount of other energy inputs, needed for example for heating, ventilation and feeding. Such inputs may have effects on the biological processes of animal (e.g. regulating environmental conditions through heating/ventilation can affect the animal heat production), but it is quite clear that there is no straightforward method to relate such inputs to the metabolic energy flow/partitioning within the animal.

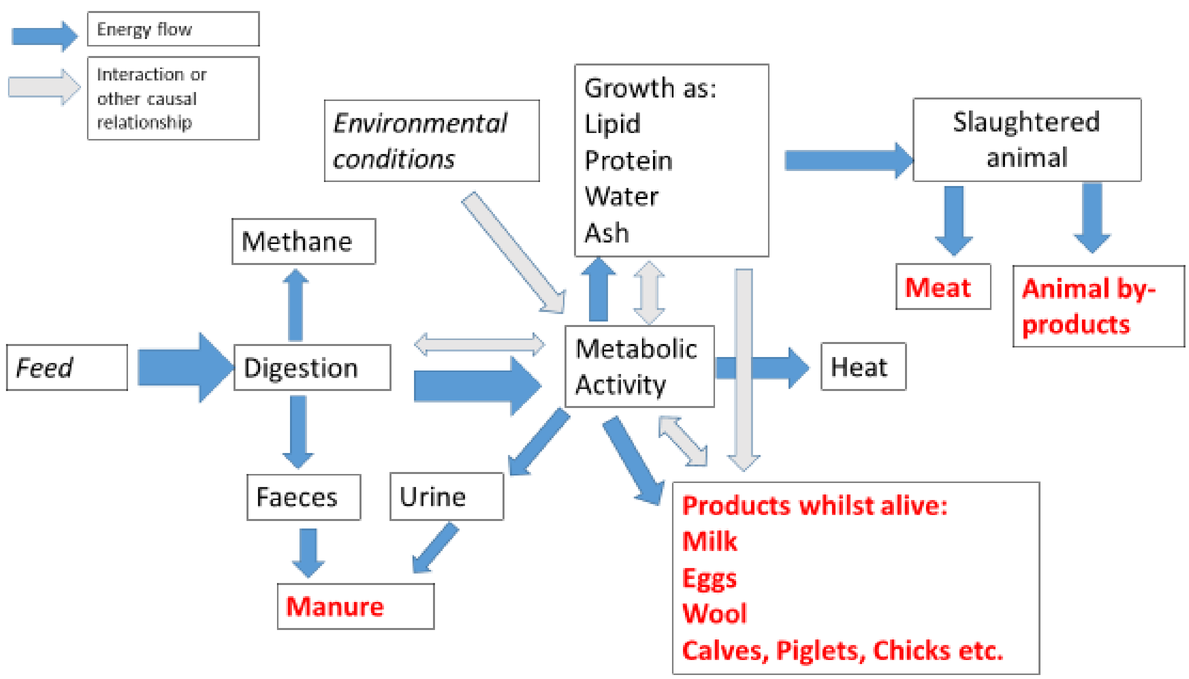


Figure 2.2 A simplified schematic of energy flow and other causal relationships in animals in livestock production systems. Inputs to the system are indicated in italics while potential co-products are in red bold font.

Figure 2.3 represents a simplified schematic of the crop production system, including growth of plants, the flow of the energy within the system, its partitioning to various co-products and the complicated interactions between these processes. Unlike in the animal production systems, in crop production all energy involved in biological processes enters the system in the form of solar radiation and is then transferred to chemical energy (sugars) through the process of photosynthesis. The energy is subsequently partitioned to other compounds, including starch, lipids and proteins, which then are used to construct structural organs, which are necessary to support other functions such as formation of reproductive organs and new leaves which are required for photosynthesis. Some of these organs are readily useful for human consumption or animal feed, and some of them can also be considered as raw materials of further refined co-products such as oils and protein meals. Interestingly, the solar energy input is something that is normally not considered in agricultural LCA modelling as an accountable input to the system. Furthermore, other resources considered as inputs in LCA models, such as fossil fuels used in field operations or fertilizers which provide necessary nutrients for the crops cannot be directly linked to the physical process of energy flow and partitioning within the plants.

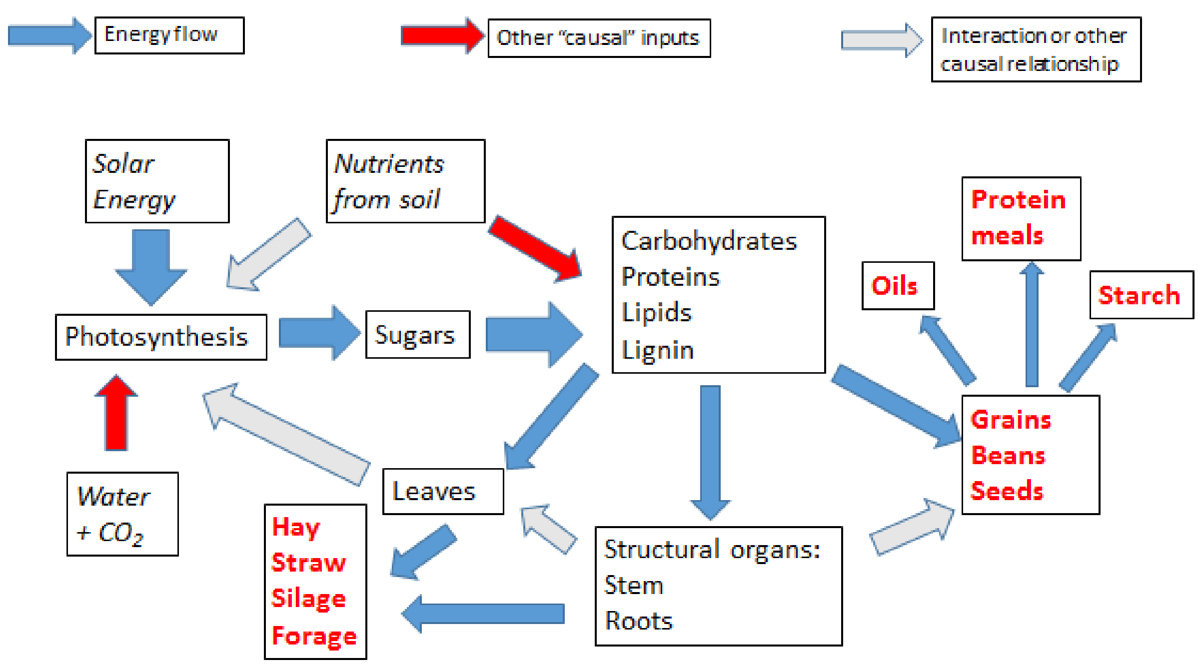


Figure 2.3 A simplified schematic of energy flow and other causal relationships in crop production. Inputs to the system are indicated in italics while potential co-products are in red bold font.

In general, the Figures 2.2 and 2.3 demonstrate the complexity of biological systems, multiple animal and plant-based co-products originating from the systems and the need for allocation of the environmental burdens between these co-products. Below we present and discuss some examples of recent attempts to solve these problems through proposed "biophysical" allocation methodologies specifically developed to address this modelling issue in agricultural systems.

2.5.1 Example 2 – allocation methodologies for co-products from dairy systems

The three outputs to which environmental impacts are generally allocated within allocation frameworks for dairy systems are meat from culled cows, meat from veal calves and milk (Gac et al., 2014). In some countries, manure from dairy systems may also be an output, but this is generally excluded from allocation frameworks for dairy farming systems. Generally these methodologies are based on defining the feed intake required by animals to produce the respective outputs which are defined as co-products. For example the International Dairy Federation (IDF) (2010) methodology proposes that the allocation factor between meat and milk is based on the empirical relationship shown in Equation 1.

Equation 1) $AF = 1 - 5.77R$

Where AF = allocation fraction for milk and R = kg beef / kg milk where kg milk is corrected to 4% fat and 3.3% protein.

This empirical equation is used to represent the feed requirement of the animal for the production of milk and meat based on the idea that "feed energy available for growth, for a given feed, is more readily available than that available for milk production". According to the authors the allocation methodology represents the "causal connection" between the feed, the major farm input and the products (International Dairy Federation, 2010). The authors acknowledge that equation 1 is empirical, but justify that it was based on data from a larger trial, which related feed intake to the "net energy content" of the feed and the subsequent production of milk and beef in a causal manner. However, the net energy value of feed is an empirical representation itself of the underlying relationships which govern the energy value of feed for animals (Ferrell and Oltjen, 2008), and was developed in order to predict feed intake and animal performance while applying many empirical adjustments for different production stages, genotypes etc. The principles of the IDF methodology have since been adopted in subsequent LCA studies of dairy systems (Dollé and Gac, 2012; Flysjö et al., 2011; Thoma et al., 2013), as well as now being recommended in the LEAP guidelines on carbon footprinting in small ruminant systems (FAO, 2014b). Thoma et al. (2013) collected data from US farms and concluded that allocation between milk and meat, based on the energy requirement to produce them, was best represented in simplified form as in equation 2:

Equation 2) $AF = 1 – 4.39R$

Where AF = allocation fraction for milk and R = kg beef / kg milk where kg milk is corrected to 4% fat and 3.3% protein. It should be noted that in this proposed framework leather is not considered as a co-product

The discrepancy between this equation and the IDF methodology arises from the empirical nature of the underlying equations and concepts being used. Similar differences are seen when applying the methodology of Dollé and Gac (2012) to French dairy systems compared to the IDF methodology, with the former allocating 73% of impacts to milk production and the latter 82%. Gac et al. (2012) consider that the animal has requirements for five functions: maintenance, activity, growth, gestation and lactation and use a mix of system separation and allocation based on energy requirement to allocate the environmental impact of animal production between meat, milk and veal calves (Gac et al., 2014). Allocation between milk and meat in dairy systems has also been carried out on the basis of the energy and protein requirement to produce meat and milk respectively (Basset-Mens et al., 2009; Cederberg and Mattsson, 2000; O'Brien et al., 2012). However, it is questionable whether "causality" is or can be demonstrated in any of these methodologies (see Figure 2.2). For example it is not possible for milk to be produced without the feed input to raise heifers into adulthood. Furthermore, any maternal growth taking place during lactation would not be possible without the energy and nutrient flow from feed to milk production (Friggens et al., 2004; Houdijk et al., 2001). Therefore, it is not possible to model causality in the system simply by modelling the flow of energy input from the feed to the various functions of the animal as these cannot be varied independently.

2.5.2 Example 3 - manure as a "co-product" in layer systems

Recent FAO guidelines on the environmental impact modelling in livestock systems specifically advise that manure is considered a co-product in cases where it has economic value (FAO, 2014b, 2014c). Here we examine the example of allocation in an egg production system provided in the LEAP guidelines on poultry systems (FAO, 2014c). In this example, the overall environmental impacts were allocated between three co-products: eggs, spent hens for slaughter and manure sold to a nearby power plant to be used as fuel. The burdens of the production system were allocated to these three product streams based on the amount of feed (or feed energy) proposed to be consumed for each stream.

As a starting point, the guidelines use the equation specified by the National Research Council (National Research Council, 1994), which was originally developed to predict the metabolizable energy (ME) requirement of laying hens (Equation 3).

Equation 3) $ME = W^{0.75}(173-1.95T) + 5.5\Delta W + 2.07EM$

Where W = Hen weight (kg), T = Temperature (◦C), ΔW = body weight change (g/day) and EM = egg mass produced (g/day). Although this equation is purely empirical and aimed to be used for predictive purposes only, the FAO guidelines (FAO, 2014c) interpret its three terms to represent the energy partitioning between "maintenance", growth and egg production respectively, and suggest that these indicate the causal relationships to be used in the co-product allocation.

The "biophysical" allocation based on Equation 3 considers only two co-products, meat (represented by the "hen weight") and eggs. In order to consider the third co-product, namely manure, the guidelines (FAO, 2014c) break down the maintenance term to different components based on the proposed sources of the heat produced by the animal, and one of these components is then interpreted to represent the biophysical processes behind manure production. In these guidelines, this component is called the "heat increment of maintenance feeding" and is quantified using an empirical equation originally presented by Emmans (1994). This equation is claimed to describe the "utilization of feed energy for the purpose of processing feed into useful nutrients and creating the excreta", and describes the "heat increment of maintenance feeding" as a multiple linear function of three variables, namely faecal organic matter content, urine nitrogen content and methane production. It is likely that these variables were used in the original model by Emmans (1994) in order to have some measurable quantities that can be easily applied to predict the energy use of the animals, so the original idea was not to use them as representation of any "causal" relationships between inputs and outputs. In any case, it is very difficult to see any causality in these variables.

According to the example calculations following the above principles and shown in the FAO guidelines (FAO, 2014c), in a standard egg production system the "heat increment of maintenance feeding" is found to be 9.1% (25,276 kcal[1]) of the total ME fed in the diet (277,767 kcal). The ME required for growth and egg production is then calculated using the relevant parts of Equation 3.

[1] Some errors appear to be present in these calculations in the FAO report. The numbers provided in the report are ME for egg production = 48083 kcal, ME for growth 17778 kcal and Heat increment of maintenance feeding = 16944 kcal. However based on the information provided in that text we calculate the numbers should be as above

In the example total ME required for growth is 17820 kcal and ME for egg production 48231 kcal. Finally the allocation factors were calculated according to the relative size of these three flows of ME (which oddly appears to make up only 32% of the total ME content in the diet fed according to the numbers provided). In this example, the allocation factors of 52.8%, 19.5% and 27.7% were obtained for eggs, hens (meat) and manure, respectively.

In the FAO guidelines, it is recommended that these allocation factors are used to "assign the whole operation emissions to the three co-products". (FAO, 2014c). This last statement seems to be at odds with the principles of biophysical allocation; it is hard to understand why parts of the Life Cycle Inventory (LCI) such as direct energy use on farm should be allocated on the basis of feed energy partitioning by the birds. There would not appear to be any quantifiable biophysical relationship between these two activities (Figure 2). Furthermore, the FAO methodology demonstrates an odd situation where LCA practitioners are allocating a proportion of the impacts from feed based on the "energy required to produce" manure. However, the report acknowledges that "physiologically speaking … the purpose is to break down the feed ingredients so that they can be absorbed and used by the animal". An example calculation where a farmer is selling poultry manure for the use as fuel to generate heat or electricity allocates 27.7% of the burdens of the whole poultry production chain to the manure as a co-product (FAO, 2014c). Where manure is not considered to have any economic value, 0% of any environmental impacts associated with poultry production would be allocated to manure, despite the fact that the flow of physical inputs and outputs to and from the bird remain unchanged. In general, the case of manure as a co-product in animal production systems presented here highlights some major issues for utilising "biophysical" allocation methodologies in agricultural production systems.

2.5.3 Example 4 – "The construction cost of plants"

Across all LCA studies of livestock systems which are not based solely on grazing, allocation issues arise in compiling an LCI of the feed supply chain, and similar issues are also valid in crop production for human consumption. Recently, a new allocation methodology for co-products from crop production has been proposed, which looks to define the energy involved in the "construction" of different categories of biomass contained within a plant (Van Der Werf and Nguyen, 2015). Plant material components are categorised as carbohydrate, protein, lipid, lignin or mineral. The "construction cost" is then calculated using the following Equations (4 & 5)

Equation 4) $Cc = (-1.041+5.077*C_{om})*(1-M) + (5.325*Norg)$

Where: Cc = the total cost to produce one gram of plant biomass (g glucose/g dry weight)

C_{om} = the carbon content of the biomass (g/g dry matter)

M = the mineral content of the biomass (g/g dry matter)

Norg = the organic N content of the biomass (g/g dry matter)

And

Equation 5) $C_{om} = 0.44 * \text{carbohydrates} + 0.535 * \text{protein} + 0.774 * \text{lipids} + 0.667 * \text{lignin}$

Environmental impacts from the production of crops in the field are then allocated according to the "construction costs" of the material contained in the outputs from crop processing, such as vegetable oils and protein meals (Van Der Werf and Nguyen, 2015).

However, the examples given in the paper ignore large sections of the plant which are not classed as co-products; all other plant material except the bean, seed or grain are ignored in the methodology (Figure 3). These appear not to be considered in the methodology on the grounds that they are not economically valuable although this is not explicitly stated. Straw is not mentioned in the presentation of the methodology, but one would have to expect that this methodology would be extremely sensitive to whether straw was considered a co-product of production in the field. If so, a large proportion of the impacts resulting from crop production would be allocated to straw, the construction of which would require a high input of solar energy.

In theory, the approach presented above can be seen to describe the physical energy flow in the crops, i.e. certain amount of absorbed solar radiation is needed to produce a certain amount of glucose, which is subsequently transformed to other compounds such as carbohydrates, lipids and proteins, and in the case of the protein (or organic nitrogen, as expressed in Equation 4), a correction is made to represent the higher "construction cost" of this compound. However, it is not clear how considering only the solar energy input to plant growth can be interpreted to represent all causal input/output relations in crop production. The methodology does not model any interaction with the nutrients (or inputs as fertilizers) available from the soil or the availability of water (and potential irrigation input), both of which are potential limiting factors on crop yields (Gregory et al., 1997). Therefore, a model which could account for these inputs would need to be used in order to develop an allocation methodology which would describe the causal relationship between the actual inputs and outputs of crop production. By definition, any causal methodology would have to consider how changes in these inputs would affect the composition of the whole plant and establish how this would alter the chemical composition of grains, beans or other products. Whether such a model, with a sufficient consistency to be generally used in a variety of LCA studies for crop production, could be ever constructed remains an open question.

2.6 Discussion and conclusions

There has been an obvious need to develop a consistent co-product allocation method to account for the environmental impact of agricultural products. As discussed above, several "biophysical" allocation methodologies have been proposed by LCA practitioners. To meet their own objectives, biophysical allocation methodologies must be based on causal relationships within the system established and in practise this can be often quantified only through mathematical modelling.

In this paper, we have examined whether researchers have been able to identify underlying physical relationships between the material and energy flows in agricultural systems and their products. The biophysical allocation methodologies detailed above have not adequately explained how the physical parameters chosen in each case represent causal physical mechanisms in these systems. The premise of many recent attempts at biophysical allocation methodologies in agricultural LCA has been to define relationships which describe how inputs to agricultural systems (usually in terms of energy) are partitioned between co-products. However, such models do not necessarily reflect the system behaviour in a mechanistic way. The methodologies discussed above for plant and animal production systems do not deal with causality in the same way as LCA models of industrial processes such as that for the boron production facility in Example 1. In addition, the interconnectivity between co-products where one cannot exist without the other is often ignored. Allocation based on either physical causation or an arbitrary choice can be based on a physical parameter. Although the former option can be seen preferable as it is recommended by the ISO standards, it comes with a burden to prove how causation within the system has been modelled (Finnveden et al., 2009).

It can be argued that allocation methodologies which use arbitrary physical properties (without modelling causality) of co-products are less desirable than those using non-physical causal relationships, such as economic value (Ekvall and Finnveden, 2001). However, following the argument of Ayer et al. (2007) that "economic allocation was not appropriate for LCA of seafood production, as it did not reflect the biophysical flow of materials and energy between the inputs and outputs of the production system", many researchers have favoured allocation based on physical properties within the field of agricultural LCA (Van Der Werf and Nguyen, 2015). Despite attempts to achieve these methodological requirements, it appears that common physical properties which simply reflect a functional output of co products have commonly been used and described as biophysical allocation without justification of how they reflect causal relationships within the system modelled, as demonstrated in the examples above.

Outputs from a production process are typically defined as co-products rather than residual or waste if they have economic value (directly or indirectly). This leads to a bizarre situation where LCA practitioners justify the use of allocation methodologies based on physical properties or relationships as preferable to economic allocation on the basis that they are more "scientific", while still applying economic criteria to determine whether a mass flow is classed as a co-product. Whole sections of the mass balance in a model of an agricultural system are included in or excluded from biophysical allocation systems on the basis of economic value. We see this paradox clearly in Example 3 where on the basis of having economic value or not manure from laying hens can be allocated either 27.7% or 0% of the impacts of the system. In this sense, the biophysical allocation methodologies for agricultural systems to date do not resolve the problem of mixing socioeconomic causality with physical causality, which has been identified as a significant criticism of allocation based on economic value (Pelletier and Tyedmers, 2011). In order to resolve this, allocation methodologies based only on physical relationships in a biological system must adopt a different definition for co-products based on physical properties.

Despite its well documented disadvantages, a major advantage of economic allocation (which in fact can be considered to be based on non-physical causal relationships) is the ability to apply it with methodological consistency across models of complex systems (Eady et al., 2012). As co-products are still defined as such based on their economic value, alternative allocation methodologies may include or exclude outputs from an agricultural system which are identical in the physical sense. Due to the complex nature of the mechanisms which underpin agricultural systems and high levels of interconnectivity between their outputs, it is unlikely that modellers will be able to consistently apply the principles of "underlying physical relationships" in allocation across agricultural LCA models. Researchers should acknowledge that in many cases the choice of allocation methodology is essentially arbitrary and present this openly in cases where systems are too complex to model causal mechanisms adequately.

Chapter 3: Accounting for uncertainty in the quantification of the environmental impacts of Canadian pig farming systems

3.1 Abstract

The objective of the study was to develop a Life Cycle Assessment (LCA) for pig farming systems that would account for uncertainty and variability in input data and allow systematic environmental impact comparisons between production systems. The environmental impacts of commercial pig production for two regions in Canada (Eastern and Western) were compared using a cradle to farm gate LCA. These systems had important contrasting characteristics such as typical feed ingredients used, herd performance and expected emission factors from manure management. The study used detailed production data supplied by the industry and incorporated uncertainty/variation in all major aspects of the system including: life cycle inventory data for feed ingredients, animal performance, energy inputs and emission factors. The impacts were defined using 5 metrics – Global Warming Potential, Acidification Potential, Eutrophication Potential (EP), Non-Renewable Resource Use and Non-Renewable Energy Use, and were expressed per kg carcass weight at farm gate, EP was further separated into Marine (MEP) and Freshwater (FEP). Uncertainties in the models inputs were separated into two types: uncertainty in the data used to describe the system (α uncertainties) and uncertainty in impact calculations or background data which affects all systems equally (β uncertainties). The impacts of pig production in the two regions were systematically compared based on the differences in the systems (α uncertainties). The method of ascribing uncertainty influenced the outcomes. In Eastern systems EP, MEP and FEP were lower (P<0.05) when assuming that all uncertainty in the emission factors for leaching from manure application was β. This was mainly due to increased EP resulting from field emissions for typical ingredients in Western diets. When uncertainty in these emission factors was assumed to be α, only FEP was lower in Eastern systems (P<0.05). The environmental impacts for the other impact categories were not significantly different between the two systems, despite their aforementioned differences. In conclusion a probabilistic approach was used to develop an LCA which dealt with uncertainty in the data systematically, when comparing multiple environmental impacts measures in pig farming systems for the first time. The method was used to identify differences between

Canadian pig production systems, but can also be applied for comparisons between other agricultural systems that include inherent variation.

3.2 Introduction

The environmental impacts of livestock systems have come under increased scrutiny in recent years (Steinfeld et al., 2006), resulting in greater focus on identifying and mitigating their environmental burdens. Several recent studies have quantified the environmental impact of pork supply chains in various farming systems using multiple environmental impact categories, through a Life Cycle Assessment (LCA) methodology (e.g. Pelletier et al., 2010; Wiedemann et al., 2010; Nguyen et al., 2011; Reckmann et al., 2013). A smaller number of LCAs on pig farming systems have introduced uncertainty in their results (Basset-mens et al., 2006; Macleod et al., 2013; Thoma et al., 2011). The only previous LCA of Canadian pig production used national inventory statistics to study the carbon footprint of Canadian pig production between 1981 and 2001 (Vergé et al., 2009). The first aim was to develop a probabilistic LCA framework that could provide meaningful comparisons between two pig farming systems (and indeed any livestock systems) and assess the potential of changes to production practices to reduce the environmental burdens of Canadian pig production. The starting point for this was the methodology developed by Leinonen et al. (2012) to compare UK poultry systems. The second aim of the study was to use recent industry benchmark data to quantify the environmental impacts of typical pig production systems in Canada for multiple impact metrics through an LCA methodology. Pig farming systems in two regions of Canada; Eastern (Quebec and Ontario) and Western (Alberta, Manitoba and Saskatchewan) were compared; these regions account for ~98% of commercial pig production in Canada (Canadian Pork Council, 2014). The two regions have several differences in pork production, including the feed ingredients used in typical diets, herd performance characteristics, such as kg weight gain per kg feed intake (gain: feed) and mortality rates, as well as farm management practices such as finishing weights and manure management.

3.3 Materials and methods

3.3.1Model structure

A cradle to farm gate LCA was conducted to compare the environmental impact of pork production systems in Eastern and Western Canada. The basic framework of the LCA model is shown in Figure 3.1 The three main compartments of material flow in the Life Cycle Inventory (LCI) were the production of feed ingredients, the consumption of feed, energy and other materials for on-farm pig production and the storage and land application of manure. The LCA modelled three separate stages in the pig production system; breeding (including suckling piglets), nursery (up to ~28 kg) and grower/finisher (G/F), from nursery end to finishing weight. The functional unit of the LCA was 1 kg expected carcass weight (CW) defined as live weight at farm gate multiplied by the expected carcass yield. The expected carcass yield, assuming Canadian carcass processing practices was 0.8 +/- 0.02 (Vergé et al., 2009).

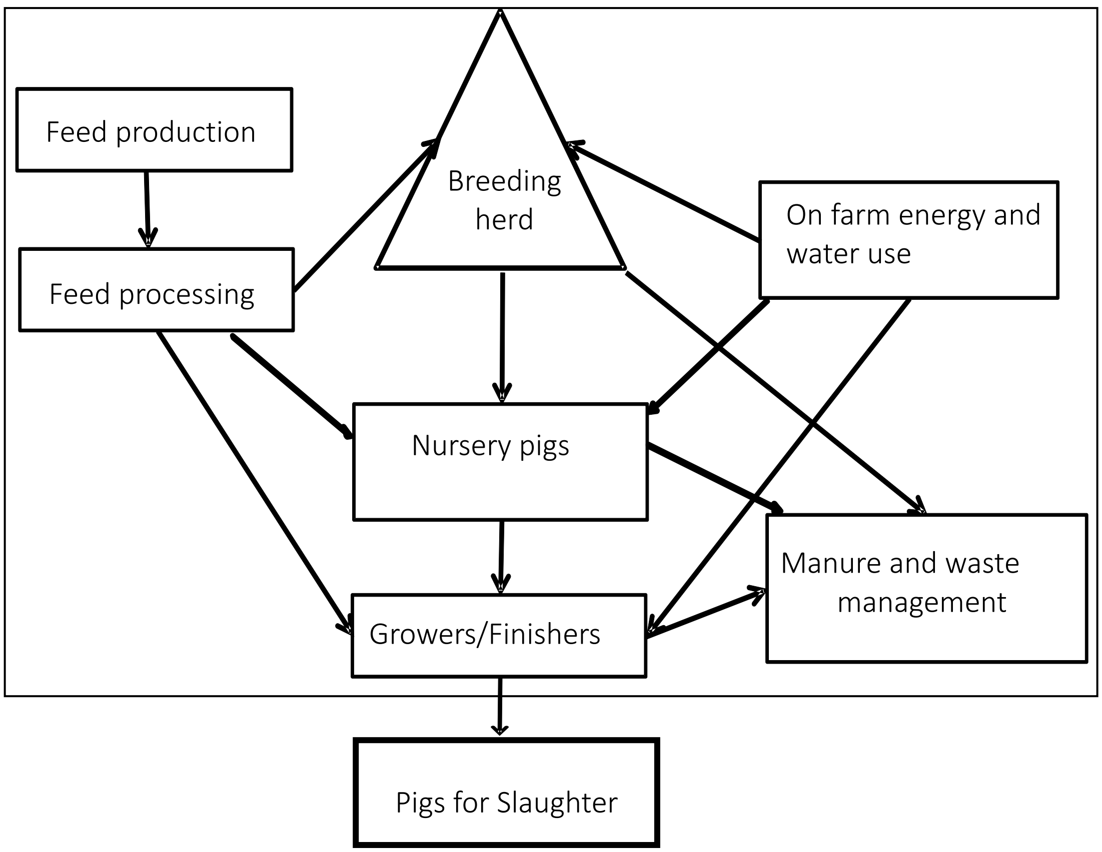


Figure 3.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment Model.

3.3.2 Impact Assessment

The environmental impacts of the systems were quantified using the following metrics: Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP), Non-Renewable Energy Use (NRE) and Non-Renewable Resource Use (NRRU). GWP was quantified as CO_2 equivalent (CO_2 eq): with a 100 year timescale; 1 kg CH_4 and N_2O emitted are equivalent to 25 and 298 kg CO_2 respectively (Intergovernmental Panel on Climate Change, 2006).

Calculations of EP, AP and NRRU followed the method of the Institute of Environmental Sciences (CML) at Leiden University (http://www.leidenuniv.nl/interfac/cml/ssp/index.html). NRE was calculated in accordance with the IMPACT 2002+ method (Jolliet et al., 2003). The EP impacts were also separated into Marine Eutrophication (MEP) for N based emissions and Freshwater Eutrophication (FEP) for P emissions using the ReCiPe midpoint method (Goedkoop et al., 2009).

3.3.3 Feed Ingredients

For both regions typical diets for each stage of production were obtained from Trouw Nutrition Agresearch: specific information on their ingredient and nutritional composition can be found in the appendix A1. The nutritional specification of these diets was in accordance with the NRC nutrient requirements recommendations for pigs (NRC, 2012). The LCI data for the major crop ingredients (barley, canola, corn, soybeans and wheat) were taken from previous LCA studies of Canadian crops (Pelletier et al., 2008; Schmidt, 2007). In Canada, > 90% of corn and 78% of soybeans are grown in the East, whereas > 90% of wheat, barley and canola is grown in the West (Statistics-Canada, 2014a). The proportional mixture of synthetic N, P and K fertilizer types was assumed to be applied to land to meet the requirements for crop production in each region (Eastern Canada for corn and soybean meal and Western Canada for wheat, barley and canola). This was derived from sales figures in a Canadian fertilizer shipments survey based on the regional breakdown of fertilizer sales (Korol, 2004). Scenarios from the Eco Invent 2.2 (Nemecek and Kagi, 2007) environmental impact database were used to model typical production processes for synthetic fertilizers to account for their environmental burdens: these scenarios were adapted to reflect the power generation mix in the Canadian electricity grid for electricity inputs (Statistics-Canada, 2013). The processing scenarios for barley, corn and wheat were adapted from milling scenarios contained in Eco Invent 2.2 to reflect Canadian energy inputs. The oil seed milling LCI data for canola and soybeans was adapted from Canadian and US scenarios respectively (Schmidt, 2007). Where system separation was not possible, co-product allocation within the feed supply chain was conducted using economic allocation, in accordance with the recommendations set out by the FAO LEAP committee (FAO, 2014a). The sources of LCI data for all other feed ingredients included in relatively small proportions in the diets can be found in appendix B.

3.3.4 Farm Model

The herd performance data used is shown in Table 3.1 and based on benchmark data from farms in Eastern and Western Canada. The Eastern data was based on the performance of 73,000 sows from 85 herds, 1.5 million nursery pigs (approx. 430 herds) and > 1 million finished pigs (approx. 470 herds). The Western benchmark data was taken from a sample of 59,000 sows (35 herds), 63,757 nursery pigs (66 herds) and 26910 finished pigs (9 herds). For the Eastern G/F pigs the average herd size was approximately 2200, in the West the average G/F herd size was 2990 (with a range of 487-4563). The overall pig population of Eastern Canada in 2011 was approximately 7 million pigs with an average herd size of 1715, in Western Canada this population was approx. 5 million with an average herd size of 3428 (Brisson, 2014). Using these data the model calculated the feed intake at each production stage per finished pig, accounting for mortality at all stages and the flow of gilts to replace culled sows.

Using the representative diets based on industry recommendations and the herd performance data shown in Table 3.1, the N, P and K retained and excreted per pig was predicted. The retention of N in the finished pigs was calculated using the principles of Wellock et al., (2004) and was assumed to be 0.0256 BW +/- 0.00128. Retention of P and K were calculated using an allometric relationship of body composition to BW (Lenis & Jongbloed, 1995; Symeou et al., 2014) and were assumed to be approx. 0.005 BW +/- 0.00025 and 0.002 BW +/- 0.0001 respectively. For K this assumption represents a linear approximation around slaughter weight of a curvilinear relationship (Rigolot et al., 2010). All N, P and K not retained by the finished pigs was assumed to be excreted in faeces or urine.

The on-farm energy consumption data was adapted from a detailed study of energy consumption in conventional pig housing systems in Iowa (Lammers et al., 2010), as there were no equivalent data for Canadian systems available. To reflect longer and colder Canadian winters in comparison to Mason City, Iowa (used in the Lammers et al. (2010) calculations), larger loads of Liquid Petroleum Gas (LPG) for heating were assumed to be required to maintain adequate barn temperatures. Based on average temperature data for Mason City (U.S. Climate Data, 2014), and regional data for Eastern and Western Canada (Weatherbase, 2014) the LPG inputs for heating barns in Eastern Canada were estimated to be 25% higher than in the Iowa case study. LPG input for heating in Western Canada was assumed to be 25% larger than for Eastern Canada. For further detail on the on farm energy inputs see appendix C. The mix of electricity generation in the LCA was the national mix for the Canadian grid (Statistics-Canada, 2013); this was assumed for all Canadian unit processes in the LCA.

Table 3.1 The mean, maximum (max) and minimum (min) values fitted from the benchmark data[1] for herd performance characteristics in Eastern and Western systems.

Indicator	East mean	East min	East max	West mean	West min	West max
G:F Nursery (g gain/g feed)	0.637	0.555	0.725	0.624	0.538	0.714
G:F Grower/finisher (g gain/g feed)	0.365	0.324	0.400	0.345	0.316	0.361
ADG Nursery (g/d)	428	335	515	455	363	515
ADG Grower/finisher (g/d)	882	752	983	836	801	953
Start weight (kg)	6.32	5.50	7.30	6.22	5.40	7.00
Weight end nursery (kg)	27.4	21.0	34.0	28.8	22.0	34.0
Finishing Weight (kg)	124	118	130	118	114	127
Average weaned litter size (piglets)	11.0	9.7	11.6	10.8	10.1	12.0
Wean to oestrus (days)	6.90	5.00	9.50	6.90	5.30	8.90
Litters /sow / year	2.45	2.20	2.55	2.43	2.30	2.50
Gestation feed/ weaned piglet (kg)	28.8	25.0	34.5	28.0	25.6	34.5
Lactation feed / weaned piglet (kg)	11.8	10.0	15.4	11.4	10.0	15.4
Creep feed / weaned (kg)	0.100	0.001	0.300	0.100	0.100	1.20
Still born	7.20%	4.00%	10.0%	7.60%	5.10%	11.9%
Post birth mortality	12.5%	6.20%	19.2%	13.6%	8.80%	20.4%
Nursery mortality	2.80%	0.64%	7.50%	3.10%	0.90%	6.00%
Grower/finisher mortality	4.00%	1.50%	9.00%	3.00%	1.80%	5.90%
Sow mortality	6.80%	3.60%	10.0%	5.00%	2.00%	9.00%
Sow culling rate	36.1%	22.0%	58.0%	41.6%	27.7%	55.6%

[1]The benchmark data presented in Table 3.1 represents farm performance data for 2012 provided by pig producers in Canada as part of a survey conducted for commercial purposes. These data represent 85 Sow herds, ~ 430 Nursery herds and ~ 470 grower/finisher herds for Eastern Canada and 35 sow herds, 66 nursery herds and 9 grower/finisher herds in Western Canada. The nature of the data provided by these farms and the large sample sizes in Eastern Canada meant it was only possible to estimate the number of herds the data represents.

3.3.5 Manure model

The manure model estimated the emissions of CH_4, NH_3, N_2O, N_2 and NOx which occurred during housing, storage and application as well as the leaching of NO_3 and PO_4. Indirect N_2O formation resulting from NH_3 and NOx emissions and NO_3 leaching were also modelled in accordance with the IPCC (2006) principles. Manure was assumed to remain in the barn for up to 7 days; it was then transferred to outside storage (except in cases where storage was a pit beneath the barn). Manure was assumed to be applied to land twice annually in spring and autumn. The model of NH_3 emissions for housing and storage was based on a previous model of NH_3 emissions from pig production in Canada (Sheppard, et al., 2010b). A tier 2 IPCC methodology was adopted for emissions of CH_4, N_2O, NOx and NO_3, but adapted to reflect small N losses at housing. As average ambient temperatures were considered to be < 0 °C during winter in both regions (Weatherbase, 2014), emissions during this period were considered negligible for outside storage methods. Leaching of NO_3 contained in manure was considered to be negligible for all storage methods except unlined lagoons. All emission factors in the manure model were adapted to use studies to reflect local conditions where possible: for the full set of emission factors and sources see appendix D. The proportional mix of floor types in pig housing, storage and application techniques in the two regions was based on information from the Livestock Farm Practice Survey contained in Sheppard et al. (2010b), as well as Statistics Canada records regarding the storage and application of swine manure (Beaulieu, 2004; Statistics-Canada, 2003).

All N, P, K excreted in faeces or urine was assumed to be applied to land as fertilizer, once losses during housing and storage were accounted for. The manure as applied to land was assumed to replace the need to apply equivalent synthetic fertilizers at a rate of 0.75, 0.97 and 1 for N, P and K respectively (Nguyen et al., 2011). The proportional mixture of the types of synthetic fertilizers replaced by the NPK content of the manure in each region was derived from sales figures for Eastern and Western Canada to assume a regional average fertilizer mix (Korol, 2004). The emission factors assumed for typical application of manure and inorganic fertilizers in Eastern and Western Canada can be found in appendix D. Airborne emission factors for NH_3 and N_2O as well as NO_3 leaching from manure & inorganic fertilizer application were considered to be higher in Eastern than Western Canada systems (Rochette et al., 2008; Sheppard et al., 2010a). This was based on the principle that the ratio of precipitation to potential evapotranspiration is much higher in Eastern than in Western provinces, and this heavily influences potential volatilisation and leaching (Rochette et al., 2008).

3.3.6 Impact Sources

The results for each impact metric were assigned to the following material (and energy) flow categories to demonstrate their relative contribution to the overall impacts:

1) Feed: production of crops and additives, feed processing and transport. This category also included the water consumed during housing.

2) Electricity: direct electricity consumption at the farm (breeding, nursery and grower/finisher stages) not including feed production, processing and transport.

3) Fuel: direct fuel consumption at the farms, not including feed production, processing and transport.

4) Housing: direct emissions in the barn

5) Manure: Burdens resulting from manure storage and application. This category also includes removed burdens from replacing synthetic fertilizers.

3.3.7 Sensitivity analysis

A sensitivity analysis was carried out on the model using the mean values for the Eastern Canadian system as a test case. The aim was to identify the largest sources of uncertainty, as well as key components in the system to which different environmental impacts were very sensitive. As the LCA was a linear model, its parameters were tested on an individual basis to the upper and lower 95% confidence intervals of their assigned distributions. The only exception to this was parameters with triangular distributions (such as finishing weight) which were tested to their maximum and minimum values. The results for these upper and lower values were then reported in comparison to the mean result of the LCA based on the percentage difference between them for each impact category. Where the distribution of a parameter tested was not normal, the difference in the results will not be equal for the upper and lower value, so the upper and lower values for each parameter are reported separately. A full list of parameters tested in the sensitivity analysis and the upper and lower values tested can be found in appendix E.

3.3.8 Uncertainty Analysis

A Monte Carlo approach was applied to quantify the uncertainties associated with the impacts in both systems (Leinonen et al. 2012); The LCA calculations and Monte Carlo simulations were conducted using the SimaPro 7.2® software package. The LCA was run 1000 times; this number of simulations was tested and the results showed high levels of repeatability and sufficiently low SEM values to compare the impacts of the two systems. During each run a value of each input variable was randomly selected from a specified distribution for this variable. Distributions were assigned to variables in the LCA based on the data available in each case. For example, for major crops, yield data from 2010 to 2014 was used to estimate the average and typical ranges in the yield. In cases where it was not possible or suitable to fit distributions, triangular distributions were assigned on the basis of the mean, maximum and minimum values available. Many of the distributions in generic unit processes taken from the ecoinvent database (Swiss Centre for Life Cycle Inventories, 2007) (e.g. transport emissions) were assumed to be log-normal.

Uncertainties in LCA calculations can be classified as either system "α" or shared calculation "β" uncertainties (Wiltshire et al., 2009): α uncertainties are those considered to vary between systems, while β uncertainties are the same for both systems and in some earlier studies they have simply been ignored (e.g. Leinonen et al., 2012). For example, variation in the herd performance parameters between the two systems, used to calculate feed intake (Table 3.1), would be considered α uncertainties. Uncertainty contained in LCI data for a key infrastructure process such as natural gas extraction is an example of β uncertainty. In the case of emission factors for manure application, there were two possible approaches arising from this methodology. This was due to the complexity of comparing systems covering large geographical areas. On average, emission factors were considered to be different for the two systems, due to differing climatic and soil conditions. However these assumptions, particularly regarding leaching of NO_3 and PO_4 were subject to a large amount of uncertainty. The emission factors thus contained both α and β uncertainty which could not be separated.

In order to assess whether the regional results were significantly different for any of the impact categories, parallel Monte Carlo simulations were run using the A>B testing function available in SimaPro 7.2®. Any process which existed in both systems was assigned the same value from its distribution of impacts to both systems for each individual comparison. This technique allowed taking the effect of β uncertainty into account in a realistic way. The output was the frequency in 1000 simulations that Eastern pig farming systems had greater or lesser impacts than Western pig farming systems for each impact category, to test whether any differences in environmental impact between the two systems was significant ($P<0.05$). Separate simulations were run classifying uncertainty in the manure and inorganic fertilizer leaching emissions for NO_3 and PO_4 as both α and β uncertainties to test the significance of this assumption.

3.4 Results

3.4.1 Impact analysis and comparison

Figures 3.2 and 3.3 show the environmental impact comparison of the two systems using parallel, comparative Monte Carlo simulations. Figure 3.2 shows these results with manure application emissions classed as β uncertainty, while Figure 3.3 classified them as α uncertainties. When uncertainties in the leaching of NO_3 and PO_4 resulting from manure and fertilizer application were considered as calculation (β) uncertainties, EP ($P<0.05$), FEP ($P<0.01$) and MEP ($P<0.01$) impacts were greater for Western pig production than Eastern pig production (Figure 3.2). When uncertainties in the leaching of NO_3 and PO_4 resulting from manure and fertilizer application were considered as system (α) uncertainties, this effect was still observed for FEP ($P<0.05$) but not observed for EP and MEP (Figure 3.3). For all other impact categories no significant difference was observed in the overall comparisons between the two systems.

The environmental impact values for the representative Eastern and Western pig production systems are in Tables 3.2 and 3.3 respectively. For all impact categories the G/F production stage accounted for >70% of impacts for both production systems. The nursery phase contributed not more than 11% for any impact category, with breeding contributing no more than 21% for any impact category. The production of feed ingredients accounted for >90% of NRRU and NRE and > 65% of GWP for both systems.

The mean AP, EP, MEP and FEP burdens observed for feed production were greater in Western than Eastern systems. This is likely due to a combination of lower feed efficiency for the G/F phase (see Table 3.1) and larger fertilizer required for the growing scenarios for wheat, which can make up a large portion of the Western diet. The average levels of N excretion predicted for the Western system were lower by around 4% in total, as protein content was lower in the typical diets (appendix A1); despite this the impact from housing emissions was almost identical in the two systems.

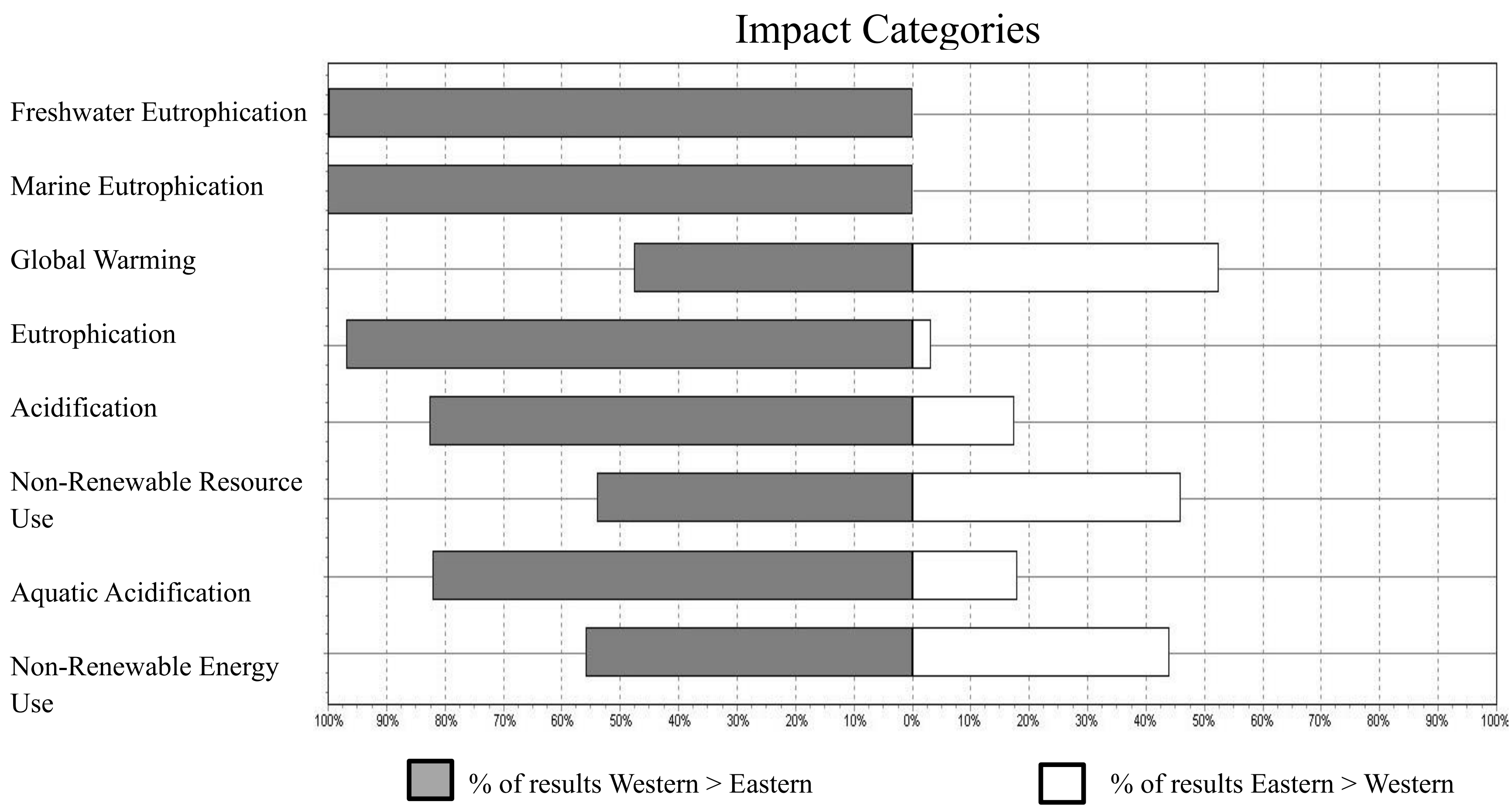


Figure 3.2 A Monte Carlo comparison of the environmental impacts (Y axis) modelled for Eastern and Western Canadian pig farming systems. The figure shows the % of instances in 1000 Monte Carlo simulations of both systems where impact of either Eastern or Western systems was greater. Any parameter which was a shared input between the two systems returned an identical value from its distribution for each individual comparison. Thus each comparison is based only on the differences between the systems. The distributions for manure application emission factors were considered as uncertainties which were shared between both systems in this case.

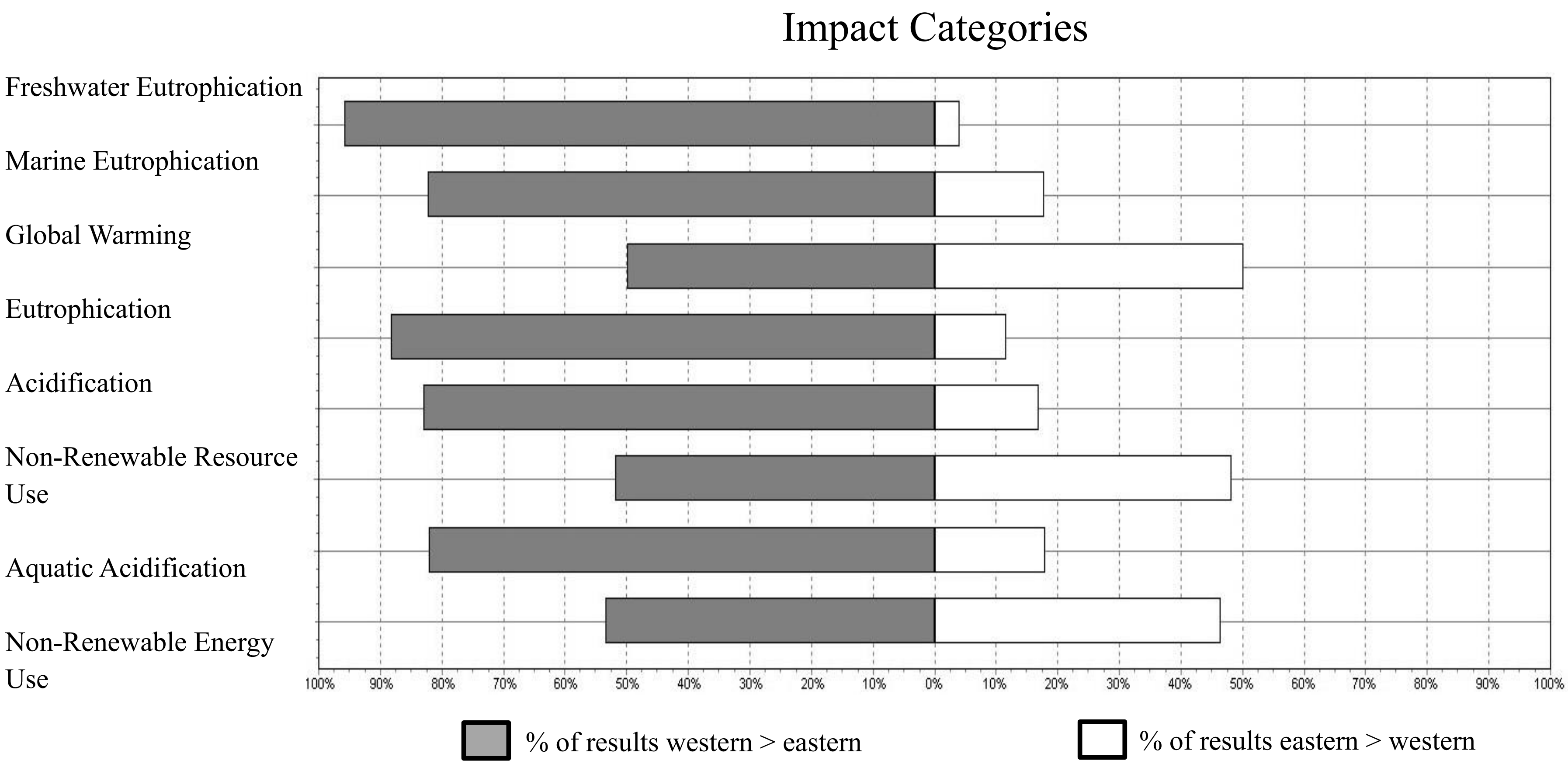


Figure 3.3 A Monte Carlo comparison of the environmental impacts (Y axis) modelled for Eastern and Western Canadian pig farming systems. The figure shows the % of instances in 1000 Monte Carlo simulations of both systems where impact of either Eastern or Western systems was greater. Any parameter which was a shared input between the two systems returned an identical value from its distribution for each individual comparison. Thus each comparison is based only on the differences between the systems. The distributions for manure application emission factors were **not** considered as uncertainties which were shared between both systems in this case.

Table 3.2 The environmental impacts of Eastern Canadian pig farming systems as outcomes of LCA; impacts are separated by source category and production stage. The mean, upper and lower values for each impact category at 95% confidence levels are shown with both α and β uncertainties considered.

	Impact Category[1,2]						
Impact source	**NRE** (MJ)	**NRRU** (kg Sb eq)	**AP** (kg SO_2 eq)	**EP** (kg PO_4 eq)	**GWP_{100}** (kg CO_2 eq)	**MEP** (kg N eq)	**FEP** (kg P eq)
Feed	19.6	0.0085	0.0190	0.0041	1.54	0.0024	0.00017
Housing	0.0	0.0000	0.0256	0.0053	0.19	0.0015	0.00000
Manure management	-3.3	-0.0015	0.0113	0.0050	0.42	0.0036	0.00014
Electricity	1.0	0.0003	0.0005	0.0001	0.04	0.0000	0.00003
Fuel	1.7	0.0008	0.0004	0.0001	0.13	0.0002	0.00000
Stage							
Breeding	3.3	0.0013	0.0080	0.0021	0.36	0.0012	0.00007
Nursery	1.6	0.0007	0.0051	0.0013	0.21	0.0007	0.00003
Grower/finisher	14.2	0.0061	0.0437	0.0109	1.75	0.0057	0.00024
Mean	19.1	0.0081	0.0568	0.0143	2.32	0.00764	0.00034
SEM	8.67E-02	4.40E-05	1.51E-04	6.29E-05	5.85E-03	1.00E-04	9.18E-06
CV	14.3%	17.2%	8.37%	13.9%	7.97%	41.5%	48.5%
Lower	14.9	0.00625	0.0481	0.0108	1.95	0.00131	5.72E-05
Upper	25.0	0.0111	0.0670	0.0181	2.69	0.0137	0.0007

[1] NRE = Non Renewable Energy Use; NRRU = Non-Renewable Resource Use; AP = Acidification Potential; EP = Eutrophication Potential; GWP_{100} = Global Warming Potential - 100 year timescale; MEP = Marine Eutrophication Potential; FEP = Freshwater Eutrophication Potential.

[2] The units for each impact category are shown in brackets, when used eq = equivalent in each case; e.g. kg PO_4 eq = kg PO_4 equivalent.

Table 3.3 The environmental impacts of Western Canadian pig farming systems as outcomes of LCA; impacts are separated by source category and productions stage. The mean, upper and lower values for each impact category at 95% confidence levels are shown with both α and β uncertainties considered.

	Impact Category[1,2]						
	NRE	**NRRU**	**AP**	**EP**	**GWP_{100}**	**MEP**	**FEP**
Impact source	MJ	(kg Sb eq)	(kg SO_2 eq)	(kg PO_4 eq)	(kg CO_2 eq)	(kg N eq)	(kg P eq)
Feed	18.5	0.0087	0.0262	0.0069	1.57	0.0074	0.00025
Housing	0.0	0.0000	0.0258	0.0054	0.18	0.0017	0.00000
Manure management	-2.9	-0.0013	0.0090	0.0046	0.35	0.0027	0.00015
Electricity	1.1	0.0003	0.0004	0.0002	0.04	0.0000	0.00003
Fuel	2.6	0.0011	0.0006	0.0001	0.18	0.0003	0.00000
Stage							
Breeding	3.5	0.0014	0.0073	0.0023	0.37	0.0019	0.00008
Nursery	2.0	0.0009	0.0066	0.0017	0.26	0.0011	0.00004
Grower/finisher	13.8	0.0059	0.0481	0.0131	1.70	0.0091	0.00031
Mean	19.3	0.00818	0.0620	0.0171	2.33	0.0121	0.000443
SEM	7.99E-02	4.25E-05	2.34E-04	6.38E-05	5.96E-03	1.11E-04	5.32E-06
CV	13.1%	16.4%	12.0%	11.8%	8.11%	29.1%	38.0%
Lower	15.8	0.007	0.0528	0.0136	1.97	0.00531	0.000152
Upper	24.4	0.0109	0.0731	0.0212	2.71	0.0193	0.000810

[1] NRE = Non Renewable Energy Use; NRRU = Non-Renewable Resource Use; AP = Acidification Potential; EP = Eutrophication Potential; GWP_{100} = Global Warming Potential - 100 year timescale; MEP = Marine Eutrophication Potential; FEP = Freshwater Eutrophication Potential.

[2] The units for each impact category are shown in brackets, when used eq = equivalent in each case; e.g. kg PO_4 eq = kg PO_4 equivalent.

The emissions of NH_3 per kg N excreted were around 4% higher in Western systems meaning the overall impacts were almost identical (see appendix D for further details on emission factors).

The combined direct electricity and fuel use on farm accounted for < 10% of all impact categories with the exception of NRRU and NRE, which were 14.1% and 13.5% respectively in Eastern systems and 19.1% and 17.1% in Western systems. Across all impact categories fuel use in Western systems caused greater impact as it was assumed a larger heating load was required to maintain temperature during a colder winter.

Manure application in both systems was associated with reducing burdens for NRRU and NRE by replacing the inputs required for the production of synthetic fertilizers. For all other impact categories the net effect of manure application was considered to increase the airborne emissions associated with these negative impacts. For all categories except NRRU and NRE impacts from manure management were on average lower in Western systems than Eastern. The manure model also predicted lower levels of total ammoniacal nitrogen volatilisation in Western systems during storage as a result of lower temperatures. Importantly emissions factors for N volatising and leaching during manure application were considered to be lower on average in Western provinces than Eastern.

3.4.2 Sensitivity analysis

Table 3.4 lists the parameters whose variability caused > 5% sensitivity for any of the impact categories tested. The magnitude of the sensitivity for each parameter is listed only for the impact categories where this was the case. Gain: feed for the G/F phase was the only model parameter for which all impact categories were sensitive to > +/-5% within its assigned variability. For NRRU, NRE and AP variability in gain: feed for G/F made this the largest source of uncertainty in the outputs. However for EP, MEP, FEP and GWP there were more sensitive parameters in the manure sub-model.

The results for EP and MEP were extremely sensitive to assumptions regarding any net difference in leaching of NO_3 caused by applying manure to land in place of inorganic fertilizer. FEP was highly sensitive to assumptions regarding the net difference between the application of inorganic fertilizer and manure for PO_4. These emission factors also contained a large amount of uncertainty which was responsible for the large CV seen in these results in Table 3.2. NRRU, NRE, GWP, EP and MEP were all sensitive > +/- 5% to the large range within the model regarding the equivalence of N in manure replacing the need for inorganic N when applied to land. Levels of AP, EP and MEP were also highly sensitive to the total ammoniacal nitrogen content of manure within a few hours of excretion.

Table 3.4 A sensitivity analysis of the parameters in the LCA, based on the typical pig production system for Eastern Canada. Parameters were tested to the upper and lower 95% confidence intervals of their distributions (except in the case of triangular distributions) in order to assess the largest sources of uncertainty in the LCA. Only cases where parameter variability caused result to move +/- 5% from mean are shown.

		Impact Category[1,2]						
Parameter	Value	**NRE**	**NRRU**	**AP**	**EP**	**GWP_{100}**	**MEP**	**FEP**
Units		(MJ)	(kg Sb eq)	(kg SO_2 eq)	(kg PO_4 eq)	(kg CO_2 eq)	(kg N eq)	(kg P eq)
Gain: feed Grow/Finisher	0.4	-5.4%	-5.5%	-8.9%	-9.1%	-6.5%	-8.7%	-7.4%
	0.324	7.9%	8.0%	13.0%	13.2%	9.4%	12.7%	10.9%
N in diet Grow/Finisher (kg N / kg feed)	0.024			-7.5%	-7.6%		-7.6%	
	0.030			7.5%	7.6%		7.6%	
P in diet Grow/Finisher (kg P / kg feed)	0.0043							-5.3%
	0.0053							5.3%
EFm_N_2O_app (kg N_2O-N/kg N applied) [3]	0.0104					-9.6%		
	0.0304					9.6%		

EFm_PO_4_app(kg PO_4-P/kg P applied) [3]	0							-23.1%
	0.04							23.1%
EFm_NO_3_app(kg NO_3-N/kg N applied) [3]	0.05				-21.9%		-92.6%	
	0.3				14.6%		61.8%	
EFs_NO_3_app (kg N-NO_3/kg N applied) [4]	0.05				16.4%		69.5%	
	0.3				-11.0%		-46.3%	
EFs_PO_4_app (kg PO_4-P/kg P applied) [4]	0							20.8%
	0.04							-20.8%
NO3_Lag (kg NO_3-N leaching / kg N unlined lagoons)	0.1						-4.4%	
	0.4						8.9%	
Inorganic N replacement rate of manure application	0.5	4.5%	4.9%		10.6%	6.8%	33.9%	
	1	-4.5%	-4.9%		-10.6%	-6.8%	-33.9%	
Manure Ammoniacal Nitrogen fraction	0.62			-8.8%	-7.4%			
	0.79			9.9%	8.3%			

[1] NRE = Non Renewable Energy Use; NRRU = Non-Renewable Resource Use; AP = Acidification Potential; EP = Eutrophication Potential; GWP_{100} = Global Warming Potential - 100 year timescale; MEP = Marine Eutrophication Potential; FEP = Freshwater Eutrophication Potential.

[2] The units for each impact category are shown in brackets, when used eq = equivalent in each case; e.g. kg PO_4 eq = kg PO_4 equivalent.

[3] EFm = emission factor manure applied to land

[4] EFs - emission factor synthetic fertilizer applied to land

GWP was more sensitive to the range of net N_2O emissions modelled for manure application than the range of gain: feed for the G/F phase.

3.5 Discussion

The objective of this study was to build a probabilistic LCA capable of quantifying the environmental impacts of current practices in the Canadian pig industry. The LCA examined the difference between pig production in Eastern (Ontario and Quebec) and Western (Alberta, Manitoba and Saskatchewan) pig production, in acknowledgement of clear differences in production practices (particularly feeding practices) in these two regions. Several large scale comprehensive LCA studies have been conducted to analyse the carbon footprint of pig farming systems e.g. (Macleod et al., 2013; Thoma et al., 2011; Wiedemann et al., 2010). Other LCA studies have also addressed different important environmental impacts which result from pig production e.g. (Basset-Mens and Van Der Werf, 2005; Cederberg and Flysjö, 2004; Nguyen et al., 2011; Pelletier et al., 2010). Only a limited number of these studies have used probabilistic modelling to account for the overall uncertainty in their findings (Basset-Mens and Van Der Werf, 2005; Macleod et al., 2013; Thoma et al., 2011) and none have done so for multiple environmental impact categories using a Monte-Carlo approach.

When comparing the environmental impacts of different production systems using LCA, it is important that uncertainty in the LCI data is accounted for to ensure confidence in the scientific rigor of any reported results (Basset-mens et al., 2006; Lloyd and Ries, 2007). While it is necessary to acknowledge all uncertainty and variability in the LCI data used for any LCA, it is also vital that any comparison made is based on the differences between two systems (Leinonen et al., 2013). LCA of animal production can be complex with 1000s of underlying unit processes being shared between any systems which are compared. As such it is important that techniques for uncertainty analysis adopted in animal LCA studies can overcome the difficulties involved in identifying where real differences exist between systems being compared. The model presented in this study was designed with these considerations in mind.

3.5.1 Uncertainty Analysis

The use of Monte-Carlo simulations allows uncertainty analysis to incorporate the complexities of different types of distribution and incorporate covariance between parameters where this is suitable (Leinonen et al., 2013). Some recent carbon footprint LCA studies of pig farming systems have recognised the importance of this and used Monte Carlo simulations to present probabilistic outputs (Macleod et al., 2013; Thoma et al., 2011).

However, when comparing the impacts of two systems in an LCA it must be recognised that a proportion of this uncertainty is shared between systems (β uncertainties) (Leinonen et al., 2013). Comparing systems based on the overlap of distributions from separate Monte Carlo simulations is not based solely on the differences between the two systems if shared uncertainty is not identified as such. This makes spotting significant differences in environmental impact less likely as LCA calculations on complex agricultural systems are likely to have a large amount of uncertainty in the overall result.

This problem was addressed by Leinonen et al. (2012) through identifying shared uncertainty and removing it completely where possible from the simulations (which simulate the systems independently) to compare the systems modelled. In cases where a process is a large input to one system and a small input to another identical system, change in the variable containing β uncertainty will not affect the systems equally. For this reason, in this study shared uncertainty was not disregarded in the overall uncertainty reported. For each comparative simulation, any unit process which input to both systems being compared was modelled identically, but each time a different value would be sampled for parameters with assigned distributions. Moreover, this LCA did not apply central limit theorem to assume normality in the impact assessment results, reflecting the fact that many of the most sensitive parameters in the LCA did not have normal distributions.

In utilising this comparative Monte Carlo approach a problem was presented when dealing with emission factors for land spreading in the manure sub-model. This was due to the challenge presented in modelling production across regions with large climatic variation. As these emission factors were very sensitive to local conditions, the potential range in these emissions was thought to be similar in both regions. However, previous studies on Canadian manure and fertilizer application showed that typically emissions for NH_3, N_2O and NO_3 leaching were lower in Western than Eastern Canada (Rochette et al., 2008; Sheppard et al. 2010a; Sheppard et al. 2010b). Thus these parameters could not be assumed to be shared inputs to both systems, but contained a large amount of shared uncertainty. Whether this uncertainty was assumed to be largely shared or not determined whether the EP burdens (as well as both MEP & FEP respectively) were greater for pig production systems in Western Canada than Eastern. This is a limitation in the case of comparing production across large regions as it is not possible to be more specific on the climatic factors which affect such emission factors. Notwithstanding this, the method is ideally suited to future application towards establishing whether changes to production practices such as changes in feeding regime can significantly reduce environmental burdens.

In this study industry-supplied benchmark data from 2012 was used to model typical farm performance for key animal performance parameters such as litter sizes, mortality and gain: feed. The large sample size for the benchmark data in Eastern Canada represented approximately 14% of all finished pigs in this region. For Western Canada a much smaller sample of benchmark data was available for use in this study for nursery and G/F pigs, representing <1% of finished pigs. The average G/F herd size of 2990 in the Western data was comparable with the 3420 reported for Western provinces in a 2011 farm survey (Brisson, 2014) and the data covered a range of farm sizes. The smaller sample size for benchmark data in Western Canada is acknowledged as a limitation of this study. However, it seems unlikely that a larger sample size, which would probably contain larger variability, would alter the conclusion that there were few significant differences between the overall impact levels for most impact categories in the two systems.

Estimation of both variation and covariation between parameters of farm performance represents a challenge when modelling uncertainty in a livestock LCA. In this study the input parameters (as listed in appendix E) were assumed to have no covariation in the model. When considering animal performance indicators in particular, this may lead to an overestimation in the variability modelled in the system, Quantifying such covariance is difficult, even in relatively obvious cases such as the correlation between gain: feed and slaughter weight (Leinonen et al., 2014). This is a modelling challenge which should be addressed in future livestock LCA

3.5.2 Sensitivity analysis

The impact analysis of pig farming systems in this LCA was further divided into 5 categories of impact source (feed, housing, manure management, electricity and fuel), as well as 3 stages of production (breeding, nursery and G/F). For Western systems feed production was the largest contributor to all impact categories; in Eastern systems this also was the case for GWP, NRRU, NRE and FEP. This is similar to findings from European pig LCA regarding the contribution of feed production to GWP (Basset-Mens and Van Der Werf, 2005; Reckmann et al., 2013). However previous studies for pig farming systems in the US and Australia have suggested that feed production may not be the largest contributor to GWP, with emissions from manure management being the largest contributor in warmer climates (Pelletier et al., 2010; Thoma et al., 2011; Wiedemann et al., 2010). Over 70% of all impacts modelled in this study for both systems were the result of activities at the grower/finisher production stage. Reducing the impact of the G/F diets would be very effective in reducing the overall environmental impact of these systems, provided that nutrient retention is not compromised to the extent of equivalent increases in emissions from housing and manure.

A parameter sensitivity analysis of the model was conducted using the input data for Eastern Canada, to identify areas of focus for further model development. As the LCA model itself only contained linear relationships, the simple analysis which tested parameters on an individual basis was suitable for identifying the largest sources of uncertainty in the model. The sensitivity analysis identified 11 parameters in the model containing uncertainty which affected the results for any impact category > +/-5%. Of these, 3 were direct properties of the production system and 8 were emission factors and assumptions which formed part of the manure model.

The FEP and MEP calculations (as well as EP to a lesser extent) were very sensitive to the assumptions made regarding emission factors for nitrate and phosphate leaching. The important assumption in this case was whether manure application for the same climatic conditions caused any net increase or decrease in the levels of leaching compared to application of the equivalent synthetic fertilizer. Conflicting evidence from previous studies on this subject (Bouwman et al., 2002; Jiao et al., 2004) meant the model was programmed to allow for both eventualities. The sensitivity to this assumption caused the large range in results for FEP and MEP shown for Eastern and Western systems.

The only variable in the LCA to which all impact categories were sensitive > 5% was the G/F gain: feed. The variability in all performance characteristics of farms modelled in this LCA was based on industry benchmark and reflects this important reality. All impact categories are sensitive to the G/F gain: feed as the inputs to feed production accounted for a significant proportion of all impact categories presented in this LCA. The importance of model variation in the impacts of major ingredients through variation in yield was tested in this analysis (see appendix E). However, the sensitivity analysis conducted here did not model alternative diets to the representative diets used and this will be an important consideration for future work. While previous LCA studies of pig farming systems have conducted scenario analysis to test the effect of alternate diets on their environmental impacts (e.g. Basset-Mens and Van Der Werf, 2005; Eriksson et al., 2005; Garcia-Launay et al., 2014) none have used stochastic LCA models to systematically account for the uncertainty in their findings.

Levels of AP and EP were sensitive to variability in the dietary content of N and P. It could be argued that for the diets presented in this study +/- 10% is an over prediction of variability for the dietary N and P contents. However, variability in feed characteristics is an important factor when modelling performance in pig farming systems which an LCA should account for. This represents an important area for future development of this model, particularly in light of the increased use of “non-traditional” feed ingredients in pig diets which may have higher levels of variability in their content (Woyengo et al., 2014).

Due to the approximate nature of the on farm energy use figures used in this study, these parameters were assigned relatively high levels of variability. No impact category tested in this study was sensitive >+/- 5% to this variability as the contribution of on farm energy use to the overall impacts of the system was < 20% for all impact categories.

3.5.3 Comparison with previous LCA studies

The environmental impacts (particularly GWP) of pig production systems in many different countries have been quantified in previous studies, and summarised recently in the FAO report on Greenhouse Gases (GHGs) which result from monogastric livestock production (Macleod et al., 2013). This report sets out a methodology to compare the results of pig LCA studies in order to correct for differing assumptions and scope. Following this methodology, the results quoted for the three LCA below are adjusted to represent the same functional unit and system boundaries as this study. The impacts characterised in this study were only directly compared with those found in LCA of North American pig farming systems.

The upper limit of GWP in this study (2.81 kg CO_2 eq / kg CW) for Eastern systems was lower than the only previous peer reviewed carbon footprint study of Canadian pig production (Vergé et al., 2009), which reported 2.96 kg CO_2 eq / kg CW in Eastern systems based on data from 2001. Vergé et al. (2009) also reported 2.83 CO_2 eq / kg CW for Western systems which was the upper limit of the results reported in this study. When comparing LCA results in this manner their complexity makes ascribing all the factors which cause discrepancies between findings on the same systems difficult. However, it is possible to pick out some key differences in the assumptions made in both models.

A contributing factor to the lower values reported in this LCA in comparison to Vergé et al. (2009) was the previously reported trend of GWP reduction over time in the pork industry (Boyd et al., 2012; Vergé et al., 2009) based on increasing levels of feed efficiency. The mean G/F gain: feed reported in the data used for this study were 0.364 and 0.345 for Eastern and Western systems respectively. The mean G/F gain: feed assumed by Vergé et al. (2009) for 2001 production was approximately 0.324 across the 5 major pork producing states which are accounted for in this study (Statistics-Canada, 2001). This is close to the lower limit of G/F gain: feed reported for both systems. However, as can be seen in the sensitivity analysis, a

G/F gain: feed of 0.324 in itself would only account for an increase in GWP to 2.62 kg CO_2 eq / kg CW in Eastern systems, accounting for around half the relative difference between the two results.

Another important source of difference between the two studies was the ingredient composition of the diets. In the Vergé et al. (2009) study the composition of the diets were assumed to have, on average, no more than 5% inclusion rate of "alternative" ingredients, such as mill screen or non-grain ingredients. In the G/F rations in both East and West systems used in this study >15% of the diet comprised by-products, such as corn DDGS and wheat shorts. The latter inclusion rate is more representative of the current practices of the Canadian swine industry. This is an important factor contributing to lower GWP reported here, as the co-product allocation principles used in this study mean these ingredients have greatly reduced GWP burdens associated with them in comparison with traditional grain ingredients. Both studies took similar approaches with regard to emissions for GHGs from manure management. In comparison to Vergé et al. (2009) this study assumed lower emission factors for CH_4 and N_2O during storage in line with a recent review (Liu et al., 2013), this will have also contributed toward the lower reduced GWP per kg CW reported here.

Recent LCA studies on US pork production have also reported higher GWP figures than this study for conventional pig production systems: with 3.08 kg CO_2 eq / kg CW reported by Pelletier et al. (2010) and 3.56 kg CO_2 eq / kg CW by Thoma et al. (2011). A large proportion of this difference can be ascribed to higher ambient temperature (along with other climatic factors) in the systems assessed in these studies. This means the emission factors for GHGs from manure management are much higher in these studies than in the Canadian systems assessed in this LCA. This is reflected in the fact that the GWP impacts from manure management in these studies were approximately 1.67 kg CO_2 eq / kg CW (Pelletier et al., 2010) and 1.44 kg CO_2 eq / kg CW (Thoma et al., 2011), in comparison to 0.61 and 0.53 kg CO_2 e/ kg CW from housing and manure management reported in this study for Eastern and Western systems respectively. Pelletier et al. (2010) reported an EP of 0.0159 kg PO_4 eq / kg CW for conventional Iowa pig farming systems, which was within the range from the Canadian systems reported here. It should be noted that comparing results of LCA studies in a systematic manner is not feasible where the uncertainty range in those results is not reported.

3.6 Conclusion

This study investigated the environmental impacts of commercial Canadian pig production and quantified them for the first time; using industry data, multiple impact metrics and a systematic analysis of uncertainty. Many recent LCA studies have underlined the importance of conducting scenario analysis to show the sensitivity of key methodological decisions made in LCA modelling on topics such as land use change and co-product allocation (Leinonen et al., 2013; Meul et al., 2012; Middelaar et al., 2013). The focus of this study however, was to present a methodology which could account for uncertainty (including variability) in LCI data while identifying differences in the environmental impacts of the pig farming systems modelled. Through separating uncertainty in the input data into α and β uncertainties, this study demonstrates a methodology for evaluating the differences between pig production systems and their environmental burdens in a systematic way. Using this baseline model it will be possible to assess whether future changes to Canadian pig production systems can significantly reduce the environmental burdens of these systems. Modelling potential changes to diets used in the industry and the ability of novel feed ingredients to reduce the overall environmental impact of pig farming systems will be a focus for this work.

Chapter 4: Can the environmental impact of pig systems be reduced by utilising co-products as feed?

4.1 Abstract

The implications of using co-products from the supply chains of human food and biofuels in pig diets, for the environmental impacts of Canadian pig systems were examined using Life Cycle Assessment. The functional unit was 1 kg expected carcass weight (ECW) and environmental impacts were calculated as: Acidification Potential (AP), Eutrophication Potential (EP), Global Warming Potential (GWP), Non-renewable Energy Use (NRE) and Non-renewable Resource Use (NRRU). Maximum inclusion limits which would not negatively affect animal performance were defined for: meat meal (55), bakery meal (87), corn DDGS (261) and wheat shorts (291) (numbers in brackets represent average across all feeding phases in g/kg as fed). Nutritionally equivalent grower/finisher (G/F) diets containing maximum inclusions of these co-products were formulated individually. These diets were compared to a simple control diet based on corn and soybean meal using 1000 parallel Monte-Carlo simulations. The maximum inclusion of meat meal reduced NRRU and NRE per kg ECW by 9% and 8% compared to the control ($P<0.001$), EP and AP increased by 10% and 7% ($P<0.001$), with no significant change in GWP. Maximum inclusion of bakery meal was found to reduce all environmental impacts for all categories modelled by < 5% ($P<0.001$). Maximum inclusion of corn DDGS in the G/F diets resulted in relatively large increases in NRRU (56%), NRE (48%) and GWP (16%) (all $P<0.001$). The maximum corn DDGS diet caused a mean reduction of <1% in AP ($P=0.01$) and did not significantly alter EP. Maximum inclusion of wheat shorts reduced GWP, NRE and NRRE by >10% ($P<0.001$) but did not significantly alter EP or AP. The environmental impact implications for pig farming systems of high inclusion levels of co-products in G/F diets formulated for economic goals (i.e. least cost per kg live weight gain), were also modelled for the first time. Four further G/F diets were formulated on a least cost basis at 100%, 97.5%, 95% and 92.5% of the energy density required for maximum feed efficiency. Minimum nutrient to net energy ratios were defined in the formulation rules to ensure the first limiting resource of all diets for growth was energy. The least energy dense diet contained the highest level of co-products (294 g/kg as fed) and the most energy dense diet contained the least (108 g/kg as fed). The least energy dense diet reduced NRE and NRRU by 9% ($P<0.001$) and GWP by 4% ($P=0.018$) when compared to the diet designed for maximum feed efficiency, but increased AP and EP by <1% ($P<0.001$). The other two intermediate levels of energy density followed the same pattern but the effects were not linear. The increased inclusion of co-products in G/F diets formulated for economic goals

can produce environmental impact reductions for some environmental impact categories in pig farming systems.

4.2 Introduction

The environmental impacts of livestock systems have come under increased scrutiny in recent years (Steinfeld et al., 2006), resulting in greater focus on identifying and mitigating their environmental burdens. Previous Life Cycle Assessment (LCA) studies have shown that feed production causes the majority of Global Warming Potential (GWP) (Basset-Mens and Van Der Werf, 2005; Macleod et al., 2013; Reckmann et al., 2013), Non-renewable Energy (NRE) and Non-renewable Resource Use (NRRU) (Chapter 3) resulting from pig farming systems. The majority of Acidification Potential (AP) and Eutrophication Potential (EP) caused by pig farming systems is due to emissions during manure storage and application, a direct result of the excretion of N and P by the animal (Basset-Mens and Van Der Werf, 2005; Dourmad et al., 2014; Reckmann et al., 2013). As such the ingredient and nutritional composition of the diets in pig farming systems are extremely important considerations when quantifying their environmental impacts. Due to the pressure of the animal feed supply chain on human food systems (Steinfeld et al., 2006), there is an increased interest in the use of alternative feed ingredients (co-products) in livestock diets (Woyengo et al., 2014; Zijlstra and Beltranena, 2013). However, the implications of including of such co-products in pig diets for the environmental impacts of the system, have not previously been investigated systematically.

Commercial pig diets are usually formulated for economic objectives (Ferguson, 2014). There are various economic objectives for which pig diets may be formulated; one of the most common is to minimise the cost of feed per kg live weight (LW) gain (ABN, 2014). Energy is the most expensive component of pig diets (Velayudhan et al., 2015). When formulating commercial diets optimum nutrient to energy ratios can be defined to ensure energy is the first limiting resource of the diet for animal growth. As feed prices vary, the optimal feeding strategy to minimise the cost of feed per kg LW gain will also fluctuate. When ingredient prices are relatively low, achieving optimum feed efficiency is less important when trying to minimise cost/ kg LW and the optimal solution may be diets of lower energy density (Saddoris-Clemons et al., 2011). Diets with lower energy density tend to cost less per tonne due to greater inclusions of low value co-products, such as wheat shorts or dried distillers grains with solubles (DDGS).

The first aim of this study was to use LCA modelling to investigate the effect of including specific co-products in grower/finisher (G/F) diets on the environmental impact of Canadian pig systems. The co-products investigated were meat (pork) meal, bakery meal, corn DDGS and wheat shorts in G/F diets. The second objective was to investigate the effect of reducing the energy density of G/F diets (and therefore the feed efficiency of the animals), whilst offering co-product based diets on the environmental impacts of pig systems.

4.3 Materials and methods

Experiment 1 examined the effect of including different co-products in G/F diets on the environmental impacts of Canadian pig farming systems; the inclusion of each co-product was assessed individually. Experiment 2 tested the effect of lowering the energy density of the G/F diets incrementally when formulating for least cost; reflecting the fact that commercial diets are not always formulated to maximise feed efficiency (Saddoris-Clemons et al., 2011; Ferguson, 2014).

4.3.1 The diets

Experiment 1: The co-products investigated were: meat (pork) meal, bakery meal, corn DDGS and wheat shorts. The consequences of their inclusion in G/F diets were compared individually to a control diet. The control diet was a simplified typical G/F diet for East Canadian pig systems; it contained none of the co-products tested and was based on corn/ soybean meal. The overall ingredient and nutrient composition (across all 4 feeding phases) of the diets in Experiment 1 are in Table 4.1; further details on the diet compositions for each feeding phase are in appendix A2. All G/F diets had nutritional specifications designed for optimum feed efficiency, following expert industry advice, as well as complying with NRC nutrient requirements (NRC, 2012a). All G/F diets were formulated for a 4 phase feeding programme (starter, grower, finisher and late finisher) on a least cost basis, using Canadian price data for 2013 provided by Trouw Nutrition Agresearch (unpublished data, see appendix F for the price ratios). The inclusion levels for each co-product were fixed to a maximum level in each feeding phase; for justification of the co-product inclusion levels see *4.3.2 Maximum Inclusion Levels.* The gestation, lactation and nursery diets were identical for all scenarios tested in this study, the composition of these diets can be found in appendix A1.

Table 4.1 The overall ingredient and nutritional composition (across all 4 feeding phases) of the grower/finisher diets tested in Experiment 1. The meat meal, bakery meal, corn DDGS and wheat shorts diets were the outcome of least cost formulations which included the maximum amount of these co-products. All ingredient inclusions shown in g/kg as fed; all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	**Control**	**Meat Meal**	**Bakery Meal**	**Corn DDGS**	**Wheat Shorts**
Canola Meal	168.6	151.8	171.1	61.2	68.2
Corn	727.8	702.9	645.8	567.4	487.3
Corn DDGS	0.00	0.00	0.00	260.6	0.00
Meat meal	0.00	64.6	0.00	0.00	0.00
Bakery Meal	0.00	0.00	86.60	0.00	0.00
Soybean meal	75.8	67.3	69.4	59.8	102.7
Wheat shorts	0.00	0.00	0.00	0.00	291.4
Limestone	12.2	3.44	12.04	14.2	14.2
Mono-calcium Phosphate	3.28	0.00	2.85	1.32	0.36
Lysine HCL	2.30	1.06	2.47	3.75	1.97
DL methionine	0.07	0.04	0.09	0.03	0.18
L Threonine	0.53	0.20	0.57	0.43	0.48
L Tryptophan	0.00	0.00	0.00	0.09	0.00
Canola Oil	0.00	0.00	0.00	0.00	24.3
Animal-vegetable fat blend	5.20	4.39	4.88	27.0	4.61
Additives	4.26	4.26	4.26	4.26	4.26
Resource					
Net Energy (MJ/kg)	9.81	9.81	9.81	9.81	9.81
Dig Crude Protein	13.26	14.69	13.22	13.84	13.50
Dig Arginine	0.85	0.98	0.84	0.78	0.96
Dig Histidine	0.41	0.43	0.40	0.41	0.43
Dig Leucine	1.21	1.31	1.19	1.50	1.15
Dig Lysine	0.80	0.80	0.80	0.80	0.80
Dig Methionine	0.26	0.29	0.26	0.28	0.26
Dig Phenylalanine	0.61	0.66	0.61	0.68	0.64
Dig Threonine	0.52	0.53	0.52	0.52	0.52
Dig Tryptophan	0.14	0.14	0.14	0.13	0.16
Dig Valine	0.62	0.70	0.62	0.66	0.66
Dig Cysteine	0.26	0.26	0.26	0.26	0.26
Ca	0.69	0.69	0.69	0.69	0.69
P	0.50	0.63	0.50	0.48	0.56
Dig P	0.24	0.37	0.24	0.27	0.28
K	0.58	0.57	0.58	0.65	0.76
Crude Protein	16.48	18.66	16.55	17.52	17.66

Nutritional values for all ingredients in the diets were primarily taken from the Stein Monogastric Nutrition Laboratory ingredient matrix (Stein Monogastric Nutrition Laboratory, 2014). In cases where certain values were missing (or ingredients themselves were missing from the matrix), values from the NRC feed ingredient tables (NRC, 2012b) and the Premier Nutrition Atlas (Hazzeldine, 2010) were used.

Experiment 2: The diets in Experiment 2 were designed to represent different feeding strategies pig producers may adopt to minimise feed cost per kg LW gain, as feed prices fluctuate. All diets were formulated on a least cost basis, with the inclusion of all co-products (with the exception of corn DDGS) permitted up to their maximum inclusion limits (see *4.3.2 maximum inclusion limits*). Experiment 1 showed that corn DDGS inclusion caused large increases in the environmental impacts of diets per kg of feed from some impact categories (see results), as such it was not included in experiment 2. The control diet was formulated using the same nutritional specifications as Experiment 1 and was designed for optimum feed efficiency (OP). Nutrient to net energy (NE) ratios remained greater than or equal to those of the OP diet for all subsequent diets. Further diets with specifications set at 97.5%, 95% and 92.5% the energy density of the OP diet were formulated, henceforth referred to as 0.975 OP, 0.95 OP and 0.925 OP. Energy was assumed to be the first limiting resource for growth in all diets. It was assumed that when the pigs were fed diets of reduced energy density, feed intake increased to achieve the same overall intake of NE across each feeding phase (Kyriazakis and Emmans, 1995). The overall ingredient and nutrient composition of the diets in Experiment 2 across all 4 feeding phases are in Table 4.2, with further details in appendix A2.

Table 4.2 The overall ingredient and nutritional composition (across all 4 feeding phases) of the grower/finisher diets tested in Experiment 2. The OP diet was a least cost formulation designed for Optimum Feed Efficiency. The subsequent diets shown were formulated at 97.5%, 95% and 92.5% the nutritional density of the OP diet (the 0.975 OP. 0.95 OP and 0.925 Op diets). All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	**OP**	**0.975 OP**	**0.95 OP**	**0.925 OP**
Canola Meal	150.1	130.1	93.8	58.4
Corn	642.3	663.2	592.7	543.4
Meat meal	0.63	1.42	3.97	1.64
Bakery Meal	82.2	28.3	28.3	5.95
Soybean meal de-hulled	69.4	64.6	64.4	70.3
Wheat shorts	25.9	89.4	191.2	287.2
Limestone	12.4	12.6	17.6	25.7
Mono-calcium Phosphate	2.59	1.85	0.42	0.00
Lysine HCL	2.72	2.70	2.60	2.45
DL methionine	0.12	0.11	0.13	0.16
L Threonine	0.71	0.69	0.66	0.61
Canola Oil	0.50	0.00	0.00	0.00
Animal-vegetable fat blend[1]	6.15	0.82	0.00	0.00
Additives	4.26	4.26	4.26	4.26
Resource				
Net Energy (MJ/kg)	9.81	9.56	9.32	9.07
Dig Crude Protein	12.94	12.63	12.42	12.21
Dig Arginine	0.82	0.81	0.82	0.84
Dig Histidine	0.39	0.38	0.38	0.38
Dig Leucine	1.16	1.14	1.10	1.07
Dig Lysine	0.80	0.78	0.76	0.74
Dig Methionine	0.26	0.25	0.25	0.24
Dig Phenylalanine	0.60	0.58	0.58	0.58
Dig Threonine	0.52	0.50	0.49	0.48
Dig Tryptophan	0.14	0.13	0.14	0.14
Dig Valine	0.61	0.60	0.59	0.59
Dig Cysteine	0.26	0.25	0.24	0.24
Ca	0.69	0.67	0.84	1.12
P	0.49	0.50	0.52	0.54
Dig P	0.24	0.24	0.25	0.27
K	0.58	0.60	0.65	0.70
Crude Protein	16.25	16.05	16.14	16.15

4.3.2 Maximum Inclusion Levels

The maximum levels of inclusion used for each dietary phase for all the co-products investigated in this study are in Table 4.3. These were defined (on an as fed basis) to levels where each ingredient could be included in pig diets without negatively affecting pig performance. The levels were set based on existing literature specific to the co products in question, as well as advice on current practices in commercial formulation.

Table 4.3 Maximum inclusion limits (g/kg as fed) used in each feeding phase in the grower finisher diets for the co-products investigated in this study

Stage	Meat Meal	Bakery Meal	Corn DDGS	Wheat Shorts
Starter	50	50	200	200
Grower	50	75	300	300
Finisher	75	100	300	400
Late Finisher	75	100	200	200

Meat meal refers to rendered animal material not including hair, hoof, horn, hide trimmings or manure as defined in article 5.1.6 of the Canadian 1983 Feeds Act (Government of Canada, 1983). In this case the animal material was assumed to be from rendered swine carcasses. Inclusions of between 5-7.5% meat meal in balanced G/F diets were not considered to affect feed conversion ratio (FCR) or average daily gain (ADG) performance in accordance with published guidelines (Bogges et al., 2008; Cromwell, 2006; OMAFRA, 2012a).

Bakery meal is surplus material from industrial baking processes (such as bread or cakes); after further processing it is sold as an ingredient for animal feed. It is defined under article 4.6.1 of the Canadian Feeds Act (Government of Canada, 1983). Very few published studies, with the exceptions of Almeida et al. (2011) and Rojas et al. (2014) have comprehensively investigated its use as a feed ingredient in pig diets. The amino acid profile of bakery meal is comparable to corn, although high processing temperatures may reduce its lysine availability (Almeida et al., 2011). Bakery meal also contains high levels of salt. Concerns about variability and consistency prevent greater utilization of bakery meal in commercial pig diets (Bogges et al., 2008; OMAFRA, 2012a). Due to the highly variable nature of this ingredient, maximum inclusion levels were limited to 10% to ensure there would be no effect pig performance in diets of equivalent nutritional specification.

Corn DDGS is a co-product of the process by which ethanol is produced from corn (Shurson et al., 2012), and is defined under article 5.5.9 of the Canadian Feeds Act (Government of Canada, 1983). Recent reviews (Gutierrez et al., 2014; Stein and Shurson, 2009; Woyengo et al., 2014) suggest that corn DDGS can be included in pig G/F diets at levels up to 30% in grower and finisher diet phases without negative effects on pig performance in terms of ADG and FCR. These studies assume a crude fat content of ~10% for corn DDGS and a similar NE value to corn. The carcass yield of pigs fed corn DDGS at levels over 15% in G/F diets may be reduced by up to 1% (Graham et al., 2014; Woyengo et al., 2014) because of higher gut fill. This reduction in carcass yield was applied in this study

As defined under article 4.2.17 of the Canadian feeds act (Government of Canada, 1983) wheat shorts are a co-product of wheat milling for flour in the North America. Wheat shorts contains fine bran particles, germ and a small portion of floury endosperm with crude fibre levels of <9%. Stein and Lange (2007) cite maximum inclusion levels of 10% for wheat shorts in nursery diets and 40% in finisher and sow diets without any adverse effects on performance. Results published by Stewart et al. (2013) suggested that 30% inclusion of wheat shorts in starter diets (for pigs 25-55kg LW) reduced ADG and increased FCR, although 30% inclusion during later dietary phases did not negatively affect these traits. Similar to corn DDGS large proportional inclusions of wheat shorts in G/F diets have been associated with reductions in carcass yield by up to 2% (Libao-Mercado et al., 2004); an average reduction of 1% was assumed in this study.

4.3.3 The LCA model

All environmental impact calculations in this study were conducted using an LCA model for pig systems in Canada; for a full description of the assumptions in this model refer to Chapter 3. The main details and in particular any deviations from the methods in that study are given below. The system boundaries of the LCA were cradle to farm-gate and the functional unit was 1 kg expected carcass weight (ECW). The environmental impacts of producing 1 kg of G/F feed were also calculated as part of the analysis. There were three main compartments of material flow in the Life Cycle Inventory (LCI): 1) the production of feed ingredients, 2) the consumption of feed, energy and other materials for on-farm pig production and 3) the storage and land application of manure. The latter included replacing the need to use mineral fertiliser through using manure as an organic fertiliser. The LCA modelled three separate stages in the pig production system; 1) breeding (including suckling piglets), 2) nursery (up to ~28 kg) and 3) grower/finisher (from nursery end to finishing weight). The inputs to the model reflected typical practices for pig production in Eastern Canada (provinces of Ontario and Quebec) which represents around 56% of Canadian pig production (Brisson, 2014).

4.3.4 Feed Production

The average environmental impacts per kg of ingredient for all ingredients used in the G/F diets can be found in Table 4.4. Where necessary economic allocation was used as the methodology for co-product allocation throughout the feed supply chain, as advised in the FAO LEAP recommendations (FAO, 2014a). The price ratios found in appendix G were used for the purposes of economic allocation. The corn-soybean meal based G/F diets tested in this study were typical of diets fed in Eastern Canadian pig systems and also reflective of diets more widely adopted in pig production in the USA. In Canada > 90% of corn and 78% soybeans produced are grown in Ontario and Quebec, conversely >90% of canola, wheat and barley are produced in the western provinces (Statistics-Canada, 2014a). LCI data for the production of major crops was adapted from a previous LCA on Canadian crop production (Pelletier et al., 2008). The LCI data for amino acids lysine, methionine, threonine and tryptophan was taken from Garcia-Launay et al. (2014). LCI data for the production of minerals mono-calcium phosphate, salt and limestone came from the Ecoinvent databases (Swiss Centre for Life Cycle Inventories, 2007). Corn DDGS was assumed to be sourced from Canadian bioethanol producers. LCI data for corn DDGS was adapted from data representative of ethanol production in the USA (Swiss Centre for Life Cycle Inventories, 2007) to be more reflective Canadian inputs of corn and energy. The LCI for bakery meal was based on data provided by a large retailer of bakery meal (Sugarich, personal communication) and adapted for a Canadian scenario. Surplus material from bread production is a large proportion of the material used for bakery meal that is sold for use in monogastric diets (Sugarich, personal communication). Bread was used as a representative input material to bakery meal in this study. The LCI for the production of 1 kg bread was adapted from the LCA food database (Nielsen et al., 2003) with the input of Canadian wheat and energy sources. A price ratio of 10:1 was assumed for bread and surplus material, with on average 8% of material collected as surplus from the bread supply chain; either during the production process or discarded at the supermarket (Sugarich, personal communication).

Table 4.4 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets in the scenarios tested.

Impact category [1]	**NRE**	**NRRU**	**AP**	**EP**	**GWP**
Unit[2]	**MJ**	**kg Sb eq**	**kg SO_2 eq**	**kg PO_4 eq**	**kg CO_2 eq**
Canola meal	3.2	1.39E-03	7.97E-03	1.59E-03	0.30
Canola oil	8.9	3.84E-03	2.20E-02	4.40E-03	0.84
Corn	4.0	1.71E-03	5.13E-03	1.11E-03	0.39
Soybean meal	1.3	5.70E-04	4.11E-03	8.71E-04	0.15
Wheat	4.2	1.84E-03	1.01E-02	2.04E-03	0.43
Meat (pork) meal	2.4	1.05E-03	2.46E-04	6.16E-05	0.13
Corn DDGS	13.9	6.51E-03	1.13E-03	2.66E-04	0.78
Wheat shorts	1.2	5.12E-04	2.78E-03	5.59E-04	0.12
Bakery meal	1.2	5.17E-04	1.41E-03	2.60E-04	0.08
Animal-vegetable fat blend	5.9	2.57E-03	1.01E-02	2.06E-03	0.49
HCL-Lysine	83.0	3.51E-02	2.12E-02	9.97E-03	4.81
L-Threonine	83.0	3.51E-02	2.12E-02	9.97E-03	4.81
DL-Methionine	80.5	3.64E-02	7.54E-03	1.70E-03	2.95
L-Tryptophan	166.0	7.01E-02	4.24E-02	1.99E-02	9.62
Sodium Chloride	3.1	1.21E-03	8.97E-04	6.68E-04	0.18
Mono-calcium Phosphate	21.5	9.40E-03	2.68E-02	3.63E-04	1.51
Limestone	0.4	1.31E-04	1.03E-04	3.58E-05	0.02

[1] NRE = Non-renewable energy use, NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential

[2] eq = equivalent

Processing inputs for packaging removal, drying and grinding were estimated to be 20 kWh electricity and 62 kWh natural gas per tonne of material processed (Sugarich, personal communication). LCI data for meat meal was adapted from a previous LCA study on rendering, the yields by mass from rendering 57.7% for fat and 42.3% for meat meal on average (Ramirez et al., 2012). The price ratio of rendered fat: meat meal was assumed to be 1.22 (unpublished data provided by Trouw Nutrition Agresearch see appendix G). The LCI data for wheat milling was adapted from Ecoinvent (Swiss Centre for Life Cycle Inventories,

2007) in order to represent Canadian energy inputs. Bread flour yield was estimated to be 73% on average, with remaining material flows of 2% wheat germ, 12.5% wheat shorts and 12% wheat bran (Blasi et al., 1998). A price ratio of 1:0.11:0.22:0.44 was assumed for wheat flour: wheat germ: wheat shorts: wheat bran (unpublished data provided by Trouw Nutrition Agresearch see appendix G)

4.3.5 Farm model

The baseline herd performance characteristics (FCR, litter size, mortality etc.) used in this study were the same as those modelled for pig systems in Eastern Canada in Chapter 3. The data collected represented the performance of 73,000 sows from 85 herds, 1.5 million nursery pigs (approx. 430 herds) and > 1 million finished pigs (approx. 470 herds). The retention of N in the finished pigs was calculated using the principles of Wellock et al., (2004) and was assumed to be 0.0256 BW ± 0.00128. Retention of P and K were calculated using an allometric relationship of body composition to BW (Lenis & Jongbloed, 1995; Symeou et al., 2014) and were assumed to be approx. 0.005 BW ± 0.00025 and 0.002 BW ± 0.0001 respectively. For K this assumption represents a linear approximation around slaughter weight of a curvilinear relationship (Rigolot et al., 2010). All N, P and K not retained by the finished pigs were assumed to be excreted in faeces or urine. Average expected carcass yield at farm gate was 80% (Vergé et al., 2009). For the wheat shorts and corn DDGS diets in Experiment 1, and the 0.95 OP and 0.925 OP diets in Experiment 2 this was reduced by 1%. The adjustment was made to account for increased gut fill due to the high proportion of bulky feed ingredients included in these diets (Graham et al., 2014; Libao-Mercado et al., 2004; Woyengo et al., 2014). The on-farm energy consumption data was adapted from a detailed study of energy consumption in conventional pig housing systems in Iowa (Lammers et al., 2010). To reflect longer and colder Canadian winters in comparison to Mason City, Iowa (which was used in the Lammers et al. (2010) calculations), larger loads of Liquid Petroleum Gas (LPG) for heating were assumed to be required to maintain adequate barn temperatures. Temperature data for Mason City (U.S. Climate Data, 2014), and regional data for Eastern Canada (Weatherbase, 2014) showed average annual temperatures were around 28% lower in Eastern Canada. The LPG inputs for heating barns in Eastern Canada were estimated to be 25% higher than in the Iowa case study. While this was a rough estimate, a previous sensitivity analysis showed that it was not a sensitive assumption for any of the impact categories tested here (see Chapter 3 sensitivity analysis)

4.3.6 Manure model

The manure model estimated the emissions of CH_4, NH_3, N_2O, N_2 and NOx which occurred during housing, storage and application as well as the leaching of NO_3 and PO_4. Indirect N_2O formation resulting from NH_3 and NOx emissions and NO_3 leaching were also modelled in accordance with the IPCC (2006) principles. Manure was assumed to remain in the barn for up to 7 days; it was then transferred to outside storage (except in cases where storage was a pit beneath the barn). It was assumed to be applied to land twice annually in spring and autumn. The model of NH_3 emissions for housing and storage was based on a previous model of NH_3 emissions from pig production in Canada (Sheppard, et al., 2010). A tier 2 IPCC methodology was adopted for emissions of CH_4, N_2O, NOx and NO_3, but adapted to reflect small N losses at housing. As average ambient temperatures were considered to be < 0 °C during winter (Weatherbase, 2014), emissions during this period were considered negligible for outside storage methods. The proportional mix of floor types in pig housing, storage and application techniques was based on information from the Livestock Farm Practice Survey (Sheppard et al., 2010), as well as Statistics Canada records regarding the storage and application of swine manure (Beaulieu, 2004; Statistics-Canada, 2003). All N, P, K excreted in faeces or urine was assumed to be applied to land as fertilizer, once losses during housing and storage were accounted for. The manure as applied to land was assumed to replace the need to apply equivalent synthetic fertilizers at a rate of 0.75, 0.97 and 1 for N, P and K respectively (Nguyen et al., 2011). The proportional mixture of the types of synthetic fertilizers replaced by the NPK content of the manure in each region was derived from sales figures for Eastern Canada to assume a regional average fertilizer mix (Korol, 2004). Further details on the emission factors used, as well as the proportional mix of floor types in pig housing, manure storage types and application techniques assumed are given in appendix D.

4.3.7 Environmental impact calculations

The impact categories quantified for this study were: Global Warming Potential (GWP), Eutrophication Potential (EP), Acidification Potential (AP), Nonrenewable Energy Use (NRE) and Nonrenewable Resource Use (NRRU). GWP was quantified as CO_2 equivalent: with a 100 year timescale; 1 kg CH_4 and N_2O emitted are equivalent to 25 and 298 kg CO_2 respectively (Intergovernmental Panel on Climate Change, 2006). EP, AP and NRRU were calculated using the method of the Institute of Environmental Sciences (CML) at Leiden University (http://www.leidenuniv.nl/interfac/cml/ssp/index.html). NRE was calculated in accordance with the IMPACT 2002+ method (Jolliet et al., 2003). The methodology used to account for the greenhouse gas emissions arising from land use changes followed PAS 2050 guidelines (BSI, 2011). All crops in the LCI of the feed supply chain in this study were assumed to be grown on arable land within North America that had been used for this purpose for ≥20 years, thus had no land use change-related greenhouse gas emissions associated with them. All environmental impact calculations for this study were conducted in the software package SimaPro 7.2®.

4.3.8 Uncertainty Analysis

The uncertainty analysis methodology used in this study was detailed in Chapter 3. Uncertainties were categorised as either specific to the system (α) or shared between the systems being compared (β). In Experiment 1, the co-product diets were each compared to the control diet using parallel Monte-Carlo simulations. In Experiment 2 the low energy density diets were individually compared to the OP diet in the same manner. Variation in all parameters except the G/F diet composition, feed intake during the G/F phase, nutrient excretion in the G/F phase and carcass yield were considered shared uncertainty in the comparisons. In Experiment 1 all diets met specifications designed for optimum feed efficiency, thus variation in feed intake was considered as β uncertainty. In Experiment 2 feed intake was assumed to increase as the energy density of the diets decreased, to achieve the same NE intake across each feeding phase (Kyriazakis and Emmans, 1995). However, all other variability in feed efficiency over the G/F phase was assumed to be intrinsic to the animal and its environment. This was modelled as shared uncertainty independent of the diet. Further details on the mean values and uncertainty ranges adopted for specific parameters within the model are provided in appendix E.

4.4 Results and discussion

4.4.1 Experiment 1

The consequences of the individual co-product inclusions in G/F diets on the average environmental impacts for the production of 1 kg of feed are in Table 4.5. The environmental impact results of the diets tested in Experiment 1 modelled per kg ECW from cradle to farm-gate are in Table 4.6.

Table 4.5 The average levels of environmental impact **per kg of feed** for grower/finisher diets tested Canadian pig production. The meat meal, bakery meal, corn DDGS and wheat shorts diets were least cost formulations which included the maximum amount of these co-products

Impact Category[1]	Control	Meat Meal	Bakery Meal	Corn DDGS	Wheat Shorts
Non-renewable Resource Use (g Sb eq)	1.90	1.81	1.82	3.25	1.57
Acidification Potential (g SO_2 eq)	5.71	5.30	5.32	4.46	5.03
Eutrophication Potential (g PO_4 eq)	1.22	1.14	1.16	0.98	1.08
Global Warming Potential $_{100}$ (kg CO_2 eq)	0.40	0.38	0.38	0.52	0.33
Non-renewable Energy Use (MJ)	4.49	4.27	4.27	7.32	3.70

[1] eq = equivalent

The G/F diet including meat meal had lower average values for all environmental impact categories tested in this study than the control diet per kg of feed (Table 4.5). The inclusion of meat meal reduced NRRU and NRE per kg ECW by 9% and 8% respectively in comparison to the control ($P<0.001$). However, EP and AP increased by 10% and 7% on average ($P<0.001$), with no significant change in GWP (Table 4.6). As can be seen in Table 4.1 the meat meal G/F diet contained higher levels of N (by 10%) and P (by 26%) than the control G/F diet. This was because meat meal contained higher levels of crude protein than the two main protein sources in the control diet; soybean meal and canola meal (Stein Monogastric Nutrition Laboratory, 2014). Lower digestible levels of certain amino acids (e.g. Tryptophan) in meat meal ensured it was not able to replace soybean meal or canola meal at a rate > 1 when added to the G/F diet. Therefore excretion of N and P was greater when meat meal was included in the G/F diet compared to the control, which caused the increases observed in AP and EP. Due to increased levels of nutrient excretion, no overall reduction in GWP per kg ECW was observed when comparing the meat meal diet to the control (Table 4.6). This was despite an average reduction of 5% in GWP per kg of feed (Table 4.5).

Table 4.6 The environmental impacts of **1 kg expected carcass weight** at farm gate for grower/finisher control and co-product diets tested in an LCA of Canadian pig production.. The meat meal, bakery meal, corn DDGS and wheat shorts diets were least cost formulations which included the maximum amount of these co-products. The control diet was a simple corn based diet containing none of these ingredients

Impact Category[1]		**Control**	**Meat Meal**	**Bakery Meal**	**Corn DDGS**	**Wheat Shorts**
Non-renewable Resource Use (g Sb eq)	Mean	6.52	5.95	6.36	10.2	5.28
	s.d.	0.90	0.81	0.96	1.80	1.16
	% < control[2]	N/A	100	100	0	100
Acidification Potential (g SO_2 eq)	Mean	57.4	61.6	55.8	56.5	56.9
	s.d.	4.2	5.0	4.8	4.0	4.2
	% < control[2]	N/A	0	100	99	70.8
Eutrophication Potential (g PO_4 eq)	Mean	14.4	15.8	14.1	14.3	14.6
	s.d.	1.8	2.0	1.7	1.8	1.8
	% < control[2]	N/A	0	100	56.4	15.6
Global Warming Potential $_{100}$ (kg CO_2 eq)	Mean	2.20	2.16	2.13	2.55	1.95
	s.d.	0.19	0.20	0.18	0.21	0.18
	% < control[2]	N/A	80.8	100	0	100
Non-renewable Energy Use (MJ)	Mean	15.8	14.6	15.4	23.5	12.9
	s.d.	1.9	1.7	2.0	3.7	2.0
	% < control[2]	N/A	100	100	0	100

1 eq = equivalent

2 The percentage of results (from 1000 simulations) where the impacts for the treatment diet were lower than the control diet.

The G/F diet including bakery meal had lower average impacts per kg of feed for every impact category tested than the control (Table 4.5). As well as this, the inclusion of bakery meal caused almost no change in the average N and P excretion in the system in comparison to the control. As a result the inclusion of bakery meal in the G/F diet produced small (<5% average) reductions for all impact categories tested (P<0.001) per kg ECW compared to the control (Table 4.6). Unlike for wheat shorts and corn DDGS, there is a lack of peer reviewed work defining the maximum inclusion level of including bakery meal in G/F diets without compromising pig performance. For this reason the levels of inclusion modelled in this study were conservative in comparison to guidelines on their potential inclusion limits in later stage pig diets (Bogges et al., 2008; OMAFRA, 2012a; Stein and Lange, 2007). As such the results presented here may underestimate the potential of bakery meal inclusion to reduce the environmental impacts of pig systems.

The inclusion of corn DDGS in G/F diets increased average levels of NRRU (by 71%), NRE (by 68%) and GWP (by 30%) per kg of feed compared to the control diet (Table 4.5). The increase in NRRU, NRE and GWP per kg of feed was due to the high levels of impact per kg of DDGS (see Table 4.4). The GWP levels for corn DDGS per kg of ingredient in this study were similar to values reported for US production systems using equivalent allocation methods (Kraatz et al., 2013; Thoma et al., 2011). The corn DDGS diet had lower average EP (by 22%) and AP (by 20%) per kg of feed (Table 4.5). The inclusion of Corn DDGS in the G/F diets resulted in relatively large average increases in NRRU (56%) and NRE (48%) per kg ECW as well as a 16% increase in GWP ($P<0.001$). The corn DDGS diet caused a small reduction in AP ($P=0.01$) of < 1% on average and did not significantly alter EP. Levels of N excretion were higher for the DDGS diet compared to the control due to increased dietary N content, although P excretion was slightly reduced (Table 4.1). As a result only a very small reduction was observed in AP for the DDGS diet, with no change in levels of EP per kg ECW. The inclusion of corn DDGS in pig diets increased GWP per kg ECW and this was in agreement with previous results published by Thoma et al. (2011).

When calculated per kg of ingredient wheat shorts had the lowest levels of NRRU, NRE and the second lowest GWP of the co-products investigated in this study (Table 4.4). Wheat shorts also had the highest overall inclusion levels of any of the feed co-products in G/F diets (Table 4.1). Average levels of AP and EP per kg of feed were also lower for the wheat shorts diet by 12% and 13% respectively when compared to the control diet (Table 4.5). The consequence of this was that of the co-products tested, the maximum inclusion of wheat shorts produced the largest reductions in NRRU (19%), NRE (19%) and GWP (12%) respectively per kg ECW ($P<0.001$). The inclusion of wheat shorts at these levels in G/F diets did not significantly affect the AP or EP of the system (Table 4.6). Increased N and P excretion caused by the wheat shorts diet meant AP and EP from the manure management system actually increased, offsetting the decrease in AP and EP per kg of diet. This meant there was no significant difference in the result per kg ECW for these impact measures.

4.4.2 Experiment 2

Table 4.7 shows the environmental impacts for 1kg ECW from cradle to farm gate for the diets tested in Experiment 2, when the energy density of the G/F diets was reduced on a sliding scale. Each incremental reduction of energy density in the diets tested in Experiment 2 increased the combined inclusion of co-products (wheat shorts, bakery meal and meat meal), although this increase was not linear. The OP diet contained 108 g/kg co-products, the 0.975 OP diet 119 g/kg, the 0.95 OP diet 223 g/kg and the 0.925 OP diet 294 g/kg combined co-products. As such the linear reduction of energy density in G/F diets did not have a linear effect on the environmental impacts of the system.

Table 4.7 The environmental impact of 1 kg **expected carcass weight** at farm gate for grower/finisher diets Canadian pig production. The OP diet was a least cost formulation designed for Optimum Feed Efficiency. The other three diets shown were formulated at 97.5%, 95% and 92.5% the energy density of the OP diet (the 0.975 OP. 0.95 OP and 0.925 OP diets).This allow for a higher inclusion of co-products in these diets

Impact Category[1]		**OP**	**0.975 OP**	**0.95 OP**	**0.925 OP**
Nonrenewable Resource Use (g Sb eq)	Mean	6.42	6.38	6.02	5.85
	s.d.	0.91	0.88	0.81	0.91
	% < OP[2]	N/A	12.4	100	100
Acidification Potential (g SO_2 eq)	Mean	56.1	56.5	56.8	56.2
	s.d.	4.3	4.1	4.3	4.4
	% < OP[2]	N/A	0	0	0
Eutrophication Potential (g PO_4 eq)	Mean	14.2	14.3	14.6	14.4
	s.d.	1.7	1.8	1.8	1.8
	% < OP[2]	N/A	0	0	0
Global Warming Potential $_{100}$ (kg CO_2 eq)	Mean	2.16	2.16	2.13	2.08
	s.d.	0.19	0.20	0.19	0.19
	% < OP[2]	N/A	0	86.6	98.2
Nonrenewable energy use (MJ)	Mean	15.5	15.5	14.6	14.2
	s.d.	1.9	1.9	1.7	1.9
	% < OP[2]	N/A	1.8	100	100

1 eq = equivalent

2The percentage of results (from 1000 simulations) where the impacts for the treatment diet were lower than the OP diet.

When compared to the OP diet the 0.975 OP diet increased AP ($P<0.001$), EP ($P<0.001$), GWP ($P<0.001$) and NRE ($P=0.018$) with average increases of <1% in all cases. NRRU was not significantly different between the OP and 0.975 OP diets.

The 0.95 OP diet caused average reductions of 4% and 6% for NRE and NRRU respectively relative to the OP diet ($P<0.001$). AP and EP for the 0.95 OP diet increased by 1% and 3% on average in comparison to the OP diet ($P<0.001$). There was no significant difference in GWP between the 0.95 OP and OP diets.

Compared to the OP diet, the 0.925 OP diet reduced average levels of both NRE and NRRU by 9% ($P<0.001$) and reduced GWP by 4% ($P=0.018$) per kg ECW. The 0.925 OP diet caused marginal average increases of <1% and 1% for AP an EP respectively ($P<0.001$) compared to the OP diet.

All G/F diets of reduced energy density tested in Experiment 2 increased levels of EP and AP when compared to the OP diet. As all diets had similar contents of crude protein and P to the OP diet (Table 4.2), this combined with incremental reductions in feed efficiency resulted in a linear increase in the levels of N and P excretion. However, the observed increase in these two impact categories was not linear as feed efficiency declined, with average AP and EP levels lower for the 0.925 OP diet than the 0.95 OP diet. In Experiment 1 increased inclusions of meat meal, bakery meal and wheat shorts in G/F diets all reduced the AP and EP per kg of feed, with wheat shorts causing the largest reduction (Table 4.6). The high levels of co-product inclusion in the 0.95 OP and 0.925 OP diets largely offset the increases in N and P excretion, meaning only relatively small increases in EP and AP were observed compared to the OP diet. The reduced GWP per kg feed in the 0.925 OP diet (due to the high levels of wheat shorts) compared to the OP diet, resulted in an overall reduction in GWP per kg ECW. This was despite the reduction in feed efficiency and increased N and P excretion.

The results in Table 4.7 show that formulating for optimum feed efficiency only minimised the environmental impact of the pig farming system for 2 of the 5 impact categories considered. The increased inclusion of co-products with low environmental impacts in the least energy dense diet resulted in reductions in GWP, NRE and NRRU per kg ECW; even when reduced feed efficiency and the effect of increased N and P excretion on the manure management system were accounted for.

4.5 General Discussion

Concerns over food security mean there is increased pressure on commercial animal production systems to use less human edible feedstuffs in animal feed (Steinfeld et al., 2006). Co-products from the human food supply chain and biofuel industry, not suitable for human consumption, represent a means of reducing the amount of human edible food contained in animal feed. The use of such co-products in commercial pig diets has increased in recent years due to a sustained period of price increases and price volatility for traditional cereal grains and protein meals (Woyengo et al., 2014). While the benefits of using co-products in pig diets in improving sustainability of the system are clear from an economic and social perspective, the implications for the environmental impact of the system are less so. As such, Experiment 1 represented an important step to quantify the environmental implications for including specific co-products in G/F diets using a representative LCA model of Canadian pig production. Previous LCA studies that investigated the effect of altering the ingredient composition of G/F diets on the environmental impacts of pig farming systems have mainly focussed on two areas: 1) the impact of crystalline amino acid supplementation (Garcia-Launay et al., 2014; Mosnier et al., 2011a; Ogino et al., 2013) and 2) the use of alternative protein sources to replace soybean meal in European systems (Eriksson et al., 2005; Meul et al., 2012; van Zanten et al., 2015). Meul et al. (2012) also investigated the effect of maximising co-product inclusion on the carbon footprint of European pig diets (per kg feed), but did not investigate the co-products included in this study. The implications for the environmental impacts of pig systems when specifically including meat meal, bakery meal or wheat shorts in G/F diets have not previously been presented in an LCA to our knowledge.

The results from Experiment 1 highlight the importance of including nutrient excretion and manure management in any assessment of the environmental impact of feed choice in livestock systems. If Experiment 1 only considered the environmental impacts of the feed production chain, its conclusion would have been that increased inclusions of meat meal, bakery meal and wheat shorts individually in iso-energetic diets reduced all environmental impact categories tested (Table 4.5). As can be seen in Table 4.6 however, this was not the case when accounting for the impacts from manure management; meat meal inclusion increased AP and EP levels and wheat shorts inclusion caused no significant reduction in AP or EP. Accounting for the environmental impacts of feed production from cradle to feed mill gate is therefore not sufficient when assessing feed choices in livestock systems, even when comparing diets which are assumed to cause no differences in feed intake.

The results of LCA studies of livestock systems are sensitive to the methodological approach adopted for co-product allocation (Nguyen et al., 2011; Wiedemann et al., 2015). A hierarchy for allocation methodologies is set out in ISO 14044; this states that when allocation cannot be avoided, it should preferably be based on physical relationships between the inputs and outputs (International Organisation for Standardisation, 2006b). However, in many studies of agricultural systems (including the present one), allocation between co-products is based on the economic value of co-products, not on any functional relationships within the system (Ardente and Cellura 2012). The main reason for this is that it is not possible in many cases, to identify causal physical relationships in the biological processes behind the agricultural production. Amongst the potential non-functional shared properties such as mass, gross energy, etc., the economic value of co-products can be seen as the most direct measure of their importance in production decisions. However there are drawbacks to adopting this methodology such as the inherent variability of commodity prices (Ardente and Cellura, 2012).

Concerns regarding variability in nutritional content continue to inhibit the use of co-products in commercial pig diets (Zijlstra and Beltranena, 2013). As well as variability alternative ingredients often have a high content of at least one anti-nutritional factor, which further inhibits their potential inclusion in pig diets (Woyengo et al., 2014). There remains a knowledge gap regarding how to account for the effect of the increased levels of nutritional variability caused by high levels of co-products on animal performance. Greater understanding of the implications of this variability for animal performance would enable a more complete assessment of the environmental impacts of feed choices involving variable co-products. Without the tools to confidently predict the effect of increased nutritional variability in diets on animal performance, nutritionists will often be cautious in their recommendations for including co-products in animal diets. The risks of such variability can be partially mitigated through the regular testing of ingredients as they are brought to the mill. Near Infrared Spectroscopy can be used to this effect as long as calibration using wet chemistry has been undertaken (OMAFRA, 2012b).

Diets in commercial pig production systems are formulated for economic outcomes in most cases. When formulating for such outcomes, diets are best optimised using linear programming for a specific goal using a growth model, without formulating for a fixed nutritional specification (Ferguson, 2014). This means diets are not always formulated for optimum levels of feed efficiency (as in Experiment 1), as there is a trade-off between feed cost and feed efficiency. If nutrient to NE ratios are fixed in the diet formulation rules, then as feed prices fluctuate so will the energy density of the optimum solution for a particular economic objective. At lower ingredient prices the solution will tend towards a lower energy diet with increased inclusion of low value co-products, such as wheat shorts (Saddoris-Clemons et al., 2011). This phenomenon was represented here by formulating least cost G/F diets at 4 incremental levels of energy density. To our knowledge, no LCA of pig farming systems has investigated the consequences of reducing the energy density of G/F diets on the environmental impact of the pig farming system when formulating for least cost. Just as there is a trade-off between feed intake and feed cost in diet formulation, there is a trade-off between feed intake and resulting nutrient excretion with the environmental impact per kg of a diet in pig systems for any given impact category. Experiment 2 showed this trade-off differed between impact categories; for NRRU, NRE and GWP the least energy dense diet tested had the lowest levels of these impact categories, conversely the most energy dense diet caused the lowest levels of EP and AP.

The results of Experiment 2 also demonstrate that when accounting for multiple environmental impact categories in livestock systems, feed choices can present trade-offs between different categories of environmental impact. Eriksson et al (2005) also observed a trade-off between reducing GWP but increasing EP and AP when modelling a scenario for replacing soybean meal with peas in European pig systems. The environmental impact trade-offs associated with feed choice have not been explored extensively in the case of pig systems, due to the limited number of studies in this area. Pork production has been shown to have relatively low levels of GWP in comparison to meat production from ruminants (Williams et al. 2006; de Vries & de Boer 2010; Eshel et al. 2014). However when using other environmental impact measures such as EP, AP and NRRU the impacts of pork production have been shown to be similar to those from beef production (Williams et al. 2006; de Vries & de Boer 2010). This is an important consideration when looking at the potential of co-products to reduce the environmental impacts of pig farming systems. For instance if AP and EP are seen as the most important environmental impacts of pig farming systems, the reductions in other impact categories shown by diets with higher levels of co-products in Experiment 2, may not be seen as beneficial enough to outweigh increases in AP and EP. This study focused specifically on testing scenarios to ask whether co-products be used as feed to reduce the environmental impact of pig systems. With further integration of a LCA model to a diet formulation tool, it would be possible to formulate diets to minimise specific types of environmental impact in a more holistic manner.

4.6 Conclusions

The environmental implications for pig farming systems of relatively high inclusion levels of co-products in G/F diets formulated for economic goals were quantified. Increased inclusions of co-products; such as bakery meal and wheat shorts in G/F diets formulated for economic goals can reduce the GWP, NRE and NRRU of Canadian pig farming systems. The least energy dense diet, with the greatest inclusions of co-products reduced GWP, NRE and NRRU, but caused small increases to AP and EP (<1%) per kg ECW when compared to a least cost diet formulated for optimum feed efficiency. These results suggest an overall benefit to increasing the use of co-products in G/F diets for the environmental impact of pig farming systems. The implications of utilising meat meal, bakery meal and wheat shorts individually in G/F diets for the environmental impact of pig systems were also modelled for the first time. The inclusion of bakery meal in G/F diets of equivalent nutritional specification reduced the environmental impacts of the system for every impact category modelled. Maximum inclusion of wheat shorts in diets formulated for the same specification was shown to cause reductions in GWP NRE and NRRU of >10% with no significant effect on AP and EP. This study showed that an increased inclusion of co-products in G/F diets can reduce the environmental impact of pig farming system in some cases. These findings add to a broader aim of identifying nutritional strategies to reduce the environmental impact of pig farming systems

Chapter 5: Towards a methodology to formulate sustainable diets for livestock: accounting for environmental impact in diet formulation

5.1 Abstract

The objective of this study was to develop a novel methodology that enables pig diets to be formulated explicitly for environmental impact objectives using a Life Cycle Assessment (LCA) approach. To achieve this, the following methodological issues needed to be addressed: 1) account for environmental impacts caused by both ingredient choice and nutrient excretion, 2) formulate diets for multiple environmental impact objectives, and 3) allow flexibility to identify the optimal nutritional composition for each environmental impact objective. An LCA model based on Canadian pig farms was integrated into a diet formulation tool to compare the use of different ingredients in Eastern and Western Canada. By allowing the feed energy content to vary, it was possible to identify the optimum energy density for different environmental impact objectives, whilst accounting for the expected effect of energy density on feed intake. A least cost diet was compared with diets formulated to minimise the following objectives: non-renewable resource use, acidification potential, eutrophication potential, global warming potential and a combined environmental impact score (using the aforementioned categories). The resulting environmental impacts were compared using parallel Monte-Carlo simulations to account for shared uncertainty. When optimising diets to minimise each environmental impact category individually, reductions in the said category were observed in all cases. However, this was at the expense of increasing the impact in other categories and higher dietary costs. The methodology can identify nutritional strategies to minimise environmental impacts, such as increasing the nutritional density of the diets compared to the least cost formulation.

5.2 Introduction

In commercial pig farming systems it is typical for nutritionists to formulate diets for least cost per tonne of feed for a fixed nutritional specification (Ferguson, 2014). This is most commonly done through the use of linear programming. More recently however, sustainability objectives rather than economic ones have increasingly come into consideration in diet formulation. There has been an increased interest in the quantification and mitigation of the environmental impacts of the livestock industry (Steinfeld et al., 2006). Assessing farming operations in more ways than just their economic "bottom line" may become more important as part of efforts to improve the sustainability of livestock systems.

For pig production systems feed production and manure management are the main sources of environmental impacts (Basset-Mens and Van Der Werf, 2005; Macleod et al., 2013). Life Cycle Assessment (LCA) is a generally accepted method to evaluate holistically the environmental impact during the entire life cycle of a product or system (Guinée et al., 2002), and there are many metrics through which environmental impact can be quantified. Carbon footprint or Global Warming Potential (GWP) is the metric that has received the most attention in the recent past (Weidmann and Minx, 2008). Analyses of livestock systems using LCA have shown monogastric animal production systems cause less GWP than meat production from ruminants, whether measured per kg of product or protein produced (de Vries & de Boer 2010; Williams et al. 2006; Eshel et al. 2014). Pork production is however, associated with relatively high levels of other environmental impact categories, including Non-Renewable Resource Use (NRRU), Acidification Potential (AP) and Eutrophication Potential (EP) (de Vries & de Boer 2010; Williams et al. 2006). The production of feed is responsible for the majority of GWP (up to 65%) (Basset-Mens and Van Der Werf, 2005; Eriksson et al., 2005; Macleod et al., 2013) and NRRU (up to 90%) (see Chapter 3), resulting from pig farming systems. The majority of AP and EP caused by pig production is due to emissions during manure storage and application, as a direct result of the excretion of nitrogen (N) and phosphorus (P) by the animal (Basset-Mens and Van Der Werf, 2005; Dourmad et al., 2014; Reckmann et al., 2013). As such, the ingredient and nutritional composition of the diets are extremely important considerations when quantifying the environmental impacts of pig production systems.

The objective of this study was to develop a novel methodology which enables pig diets to be formulated explicitly for environmental impact objectives using an LCA approach, whilst not penalising animal growth. The methodology was associated with the following challenges: 1) how to account for environmental impacts caused by both nutrient excretion and ingredient choice, 2) how to formulate diets for multiple environmental impact objectives, and 3) how to identify the optimal nutritional composition of diets for different objectives. An LCA model for pig farming systems was integrated into a diet formulation tool. The LCA model was then used to quantify the potential reductions that can be made to the environmental impact of Canadian pig farming systems through explicitly optimising diets for this purpose in a diet formulation tool.

5.3 Materials and Methods

5.3.1 The system under consideration

Modern pig farming systems can be considered to have 3 distinct production phases; 1) gestation and farrowing - where piglets are produced by breeding sows, 2) the nursery or weaning phase when pigs are separated from their mother and 3) the grower/finisher (G/F) phase where pigs are fattened from around 30kg to slaughter weight (PorkCheckoff, 2009). Figure 5.1 shows the major components of this system when considered in an LCA model; from the production of feed ingredients to animals shipped from the farm gate for slaughter. There were three main compartments of material flow considered in the LCA model: 1) the production of feed ingredients, 2) the consumption of feed, energy and other materials for on-farm pig production and 3) the storage and land application of manure. Benchmark data from 2012 on Canadian pig farms showed that 78% of feed consumed per pig produced and at least 75% of the environmental impacts occurred during the G/F phase (see Chapter 3). Attention therefore was given to formulating diets only for the G/F phase of production. Diets were formulated in two scenarios for pig production systems in Eastern and Western Canada because the main ingredients used in their typical diets are not the same. Pig diets in Eastern Canada are typically based on corn similar to USA pig diets (Thoma et al., 2011), whereas pig diets in Western Canada use wheat and barley as the main cereal component/s (Patience et al., 1995).

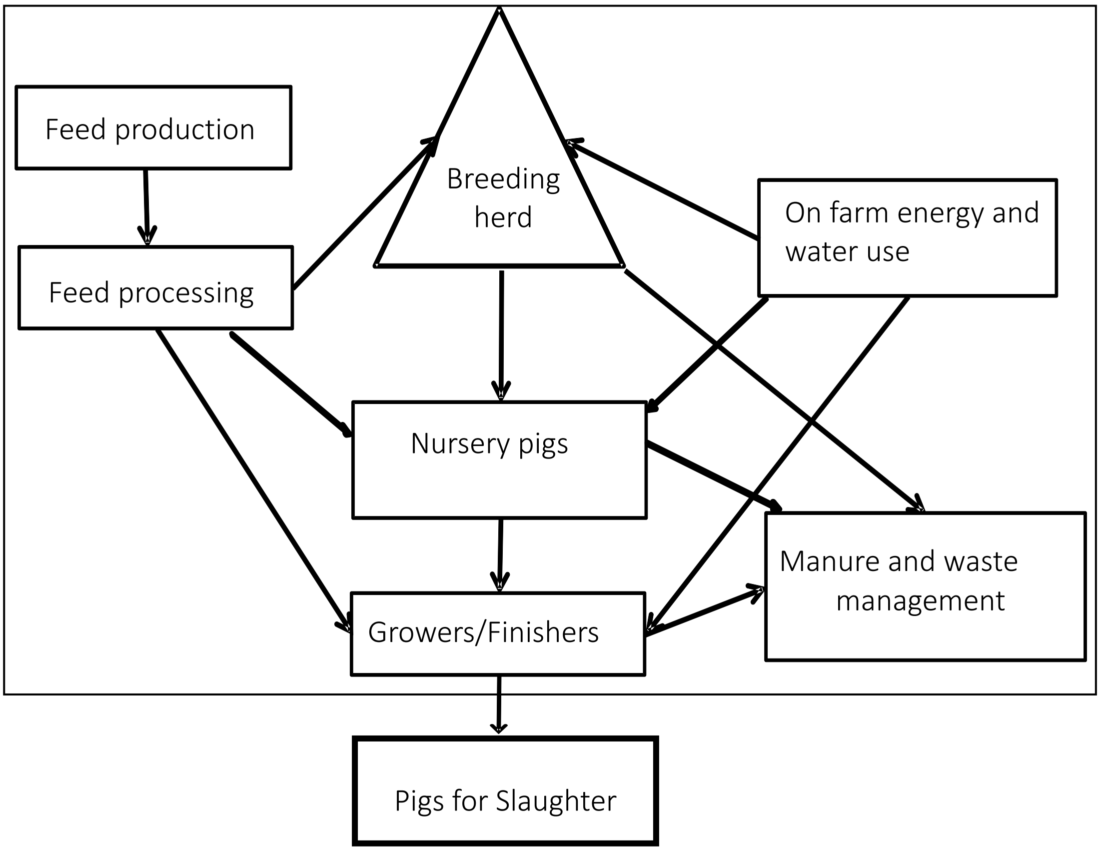


Figure 5.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment Model. Feed production in the model included the manufacture of fertilisers and pesticides etc. as inputs to growing crops.

5.3.2 The LCA model

The environmental impacts resulting from all diets formulated in this study were calculated using an LCA model of pig systems in Eastern and Western Canada (as described in Chapter 3). Some aspects of this model were also included as part of the diet formulation process (see *Diet formulation rules*). The details regarding the main components of the LCA model of Canadian pig farming systems are provided below. The system boundaries of the LCA were cradle to farm-gate and the functional unit was 1 kg expected carcass weight (ECW). The breeding and nursery production stages were treated as independent to the G/F phase in this study and remained constant for all comparisons made.

5.3.3 Feed production

The average environmental impacts per kg of ingredient for all ingredients used in the G/F diets can be found in Table 5.1. Important causes of environmental impact in the feed supply chain for pigs include: fossil fuel inputs for fertilizer production, emissions resulting from the spreading of fertilizers, fossil fuel use for field operations, energy inputs to processing (drying, grinding etc.) and transport (Van Der Werf et al., 2005). When modelling a complex supply chain, as is the case for animal feed, the inputs to the process (wheat, water, energy etc.) are shared between the different co-products resulting from these processes, and the environmental impacts associated with them must be allocated. Economic allocation was used as the methodology for co-product allocation throughout the feed supply chain as advised in the FAO Livestock Environmental Assessment and Performance partnership (LEAP) recommendations (FAO, 2014a). The price ratios found in appendix G were used for the purposes of economic allocation. Life Cycle Inventory (LCI) data for the production of major crops was adapted from a previous LCA on Canadian crop production (Pelletier et al., 2008). LCI data for amino acids; lysine, methionine, threonine and tryptophan were obtained from an LCA study on the impact of amino acids in pig diets (Garcia-Launay et al., 2014). LCI data for the production of minerals; dicalcium phosphate, salt and limestone came from the Ecoinvent databases (Swiss Centre for Life Cycle Inventories, 2007). LCI data for corn DDGS from Canadian sources was not available and therefore was adapted from data representative of ethanol production in the USA (Swiss Centre for Life Cycle Inventories, 2007) assuming the use of Canadian corn and typical electricity mix. The LCI for bakery meal was based on data provided by a large retailer of bakery meal (Sugarich, per comm, 2015) and adapted for a Canadian scenario. Surplus material from bread production is a large proportion of the material used for bakery meal sold for use in monogastric diets (Sugarich, per comm, 2015) and was used as a representative input to bakery meal in this study. The LCI for the production of 1 kg bread was adapted from the LCA food database (Nielsen et al., 2003) with the input of Canadian wheat and energy sources. A price ratio of 10:1 was assumed for bread and surplus material, with an average 10% of material collected as surplus from the bread supply chain either during the production process or discarded at the supermarket (Sugarich, per comm, 2015). Processing inputs for packaging removal, drying and grinding were estimated to be 20 kWh electricity and 62 kWh natural gas per tonne of material processed (Sugarich, per comm, 2015).

Table 5.1 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets tested. Inventory data for these ingredients was compiled as part of previous life cycle assessment studies of Canadian pig farming systems (see Chapter 3 & 4)

Impact category [1]	**NRRU**	**AP**	**EP**	**GWP**	**Combined environmental impact score** [2]
Unit [3]	**kg Sb eq**	**kg SO2 eq**	**kg PO4 eq**	**kg CO2 eq**	**<no units>**
Barley	2.18E-03	5.36E-03	2.69E-03	0.38	8.20E-14
Canola meal	1.39E-03	7.97E-03	1.59E-03	0.30	8.53E-14
Canola oil	3.84E-03	2.20E-02	4.40E-03	0.84	2.36E-13
Corn	1.71E-03	5.13E-03	1.11E-03	0.39	6.55E-14
Soybean meal	5.70E-04	4.11E-03	8.71E-04	0.15	4.33E-14
Wheat	1.84E-03	1.01E-02	2.04E-03	0.43	1.10E-13
Meat (pork) meal	1.05E-03	2.46E-04	6.16E-05	0.13	1.21E-14
Corn DDGS	6.51E-03	1.13E-03	2.66E-04	0.78	7.05E-14
Wheat Bran	1.02E-03	5.56E-03	1.12E-03	0.24	6.07E-14
Wheat shorts	5.12E-04	2.78E-03	5.59E-04	0.12	3.03E-14
Field Peas	1.32E-03	2.31E-03	2.72E-03	0.58	5.98E-14
Bakery meal	5.17E-04	1.41E-03	2.60E-04	0.08	1.73E-14
Animal-vegetable fat blend	2.57E-03	1.01E-02	2.06E-03	0.49	1.16E-13
Soybean oil	1.51E-03	1.09E-02	2.30E-03	0.40	1.15E-13
HCL-Lysine	3.51E-02	2.12E-02	9.97E-03	4.81	5.68E-13
L-Threonine	3.51E-02	2.12E-02	9.97E-03	4.81	5.68E-13
DL-Methionine	3.64E-02	7.54E-03	1.70E-03	2.95	3.71E-13
L-Tryptophan	7.01E-02	4.24E-02	1.99E-02	9.62	1.14E-12
Sodium Chloride	1.21E-03	8.97E-04	6.68E-04	0.18	2.36E-14
Dicalcium Phosphate	9.40E-03	2.68E-02	3.63E-04	1.51	2.91E-13
Limestone	1.31E-04	1.03E-04	3.58E-05	0.02	2.33E-15

[1] NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential, GWP, Global Warming Potential.

[2] Calculated by combining the total normalised NRRU, AP, EP and GWP using the CML methodology (CML, 2002) with equal weighting

[3] eq, equivalent

LCI data for meat meal was adapted from a previous LCA study on rendering, the yields by mass from rendering were assumed to be 57.7% for fat and 42.3% for meat meal (Ramirez et al., 2012). The price ratio of rendered fat: meat meal was assumed to be 1.22. The LCI data for wheat milling was adapted from Ecoinvent (Swiss Centre for Life Cycle Inventories, 2007) in order to represent Canadian energy inputs. Bread flour yields was estimated to be 73% on average, with remaining material flows of 2% wheat germ, 12.5% wheat shorts and 12% wheat bran (Blasi et al., 1998). A price ratio of 1:0.11:0.22:0.44 was assumed for wheat flour: wheat germ: wheat shorts: wheat bran. This was based on the expectation that flour would provide around 90% of the gross margin for a typical milling operation (FAO, 2009) and Canadian price data for the co-products from wheat milling as animal feed (see appendix G).

5.3.4 Manure model

The manure model estimated the emissions of CH_4, NH_3, N_2O, N_2 and NOx which occurred during housing, storage and application as well as the leaching of NO_3 and PO_4. Indirect N_2O formation resulting from NH_3 and NOx emissions and NO_3 leaching were also modelled in accordance with the IPCC principles (Intergovernmental Panel on Climate Change, 2006). Manure was assumed to remain in the barn for up to 7 days; it was then transferred to outside storage (except in cases where storage was a pit beneath the barn). It was assumed to be applied to land twice annually in spring and autumn. The model of NH_3 emissions for housing and storage was based on a previous model of NH_3 emissions from pig production in Canada (Sheppard et al., 2010b). A tier 2 IPCC methodology was adopted for emissions of CH_4, N_2O, NOx and NO_3, but adapted to reflect small N losses from housing. As average ambient temperatures were considered to be < 0 ˚C during winter (Weatherbase, 2014), emissions during this period were considered negligible for outside storage methods. The proportional mix of floor types in pig housing, storage and application techniques in each region was based on information from the Livestock Farm Practice Survey (Sheppard et al., 2010b), as well as Statistics Canada records regarding the storage and application of swine manure (Beaulieu, 2004; Statistics-Canada, 2003). All N, P, K excreted in faeces or urine was assumed to be applied to land as fertilizer, once losses during housing and storage were accounted for.

Manure applied to land was assumed to replace the need to apply equivalent synthetic fertilizers at a rate of 0.75, 0.97 and 1 for N, P and K respectively (Nguyen et al., 2011). The proportional mixture of the types of synthetic fertilizers replaced by the NPK content of the manure in each region was derived from sales figures for Eastern and Western Canada to assume a regional average fertilizer mix (Korol, 2004).

5.3.5 Farm performance

With the exception of feed intake during the G/F stage and carcass yield, the baseline herd performance characteristics (litter size, mortality etc.) used in this study were as those modelled for pig systems in Eastern and Western Canada in a previous regional LCA study (see Chapter 3). All characteristics of herd performance other than average feed intake and carcass yield were assumed to be independent of feed composition in the G/F production stage. While this represents a simplification made for the purposes of a modelling exercise it is valid for the scenarios modelled here. All diets formulated were nutritionally balanced and would not be expected to have implications for herd health status or mortality during the G/F phase. It is reasonable to expect that other model inputs such as on-farm energy use are independent of feed composition. The on-farm energy consumption data was adapted from a detailed study of energy consumption in conventional pig housing systems in Iowa (Lammers et al., 2010), as there were no equivalent data for Canadian systems available. In order to reflect longer and colder Canadian winters in comparison to Mason City, Iowa (which was used in the Lammers et al. (2010) calculations), larger loads of Liquid Petroleum Gas (LPG) for heating were assumed to be required to maintain adequate barn temperatures. Based on average temperature data for Mason City (U.S. Climate Data, 2014), and regional data for Eastern and Western Canada (Weatherbase, 2014) the LPG inputs for heating barns in Eastern Canada were estimated to be 25% higher than in the Iowa case study. LPG input for heating in Western Canada was assumed to be 25% larger than for Eastern Canada. These represent approximations as the sensitivity analysis in Chapter 3 showed that the model was not very sensitive to the assumptions made regarding LPG use for any of the impact categories tested here. The mix of electricity generation in the LCA was the national mix for the Canadian grid (Statistics-Canada, 2013); this was assumed for all Canadian unit processes in the LCA.

5.3.6 Quantifying environmental impacts

The environmental impacts of the system were quantified by the LCA using four environmental impact categories. Three of these categories quantified negative impacts resulting from emissions caused by the system; AP, EP and GWP. We included GWP as it has received the most attention in efforts to quantify the impact of livestock systems. The impact categories AP and EP were considered as they quantify the main environmental impacts which result from the storage and spreading of animal manure. The fourth impact category quantified the system's use of NRRU and was included because of the relatively high usage of cereals and oil seed meals in pig diets, which have a significant input of resources such as fertilizers (Steinfeld et al., 2006).

System GWP was quantified in CO_2 equivalents (eq) on a 100 year timescale using the IPPC methodology (Intergovernmental Panel on Climate Change, 2006). The methodology of accounting for GWP caused by land use change in this study followed the PAS 2050 guidelines (British Standards Institution, 2011). The methodologies for calculating AP (SO_2 eq), EP (PO_4 eq) and NRRU (Sb eq) were established by researchers at the Institute of Environmental Sciences at Leiden University (CML) (CML, 2002). This methodology was chosen as it is designed to quantify these impact categories on a global scale; importantly accounting for the long term impacts of airborne emissions on global levels of substances which contribute to AP and EP. The CML methodology for normalising different types of environmental impact (Huijbregts et al., 2003) was also utilised to formulate diets to minimise the combined environmental impact score of the system. The impacts which result from a process are normalised against a reference which is an estimate of the total annual level of global emissions and resource use caused by human activity (Huijbregts et al., 2003). The normalised scores for AP, EP, GWP and NRRU were then combined additively, with equal weighting to generate a combined environmental impact score in the diet formulation tool. Equal weighting was adopted in this example to ensure large increases in an individual environmental impact category did not occur when optimising to minimise the combined environmental impact score. The cradle to grave environmental impact calculations were performed in the software package SimaPro 7.3.3®.

5.3.7 Diet formulation rules

A diet formulation tool was developed which predicted the environmental impacts for each category resulting from G/F diets for the feed supply chain and manure management. The tool also quantified the feed cost per kg LW gain for each solution. The tool formulated diets using linear programming in Microsoft Excel® with the software plug in Open solver (Mason, 2011). Nutritional values for all ingredients in the diets were primarily taken from the Stein Monogastric Nutrition Laboratory ingredient matrix (Stein Monogastric Nutrition Laboratory., 2014). In cases where certain values were missing (or ingredients themselves were missing from the matrix), values from the NRC 2012 feed ingredient tables (NRC, 2012b) and the Premier Nutrition Atlas (Hazzeldine, 2010) were used. All of the G/F diets were formulated with four feeding phases (starter, grower, finisher and late finisher); this reflected typical feeding programs adopted by commercial pig operations in Canada.

The predicted start weight of the pigs in the diet formulation tool was fixed at 27.4 kg with a finish weight of 124 kg for the G/F phase, based on benchmark data collected for the LCA study of Canadian pig farming (see Chapter 3). Diets were not formulated for a fixed nutritional density, rather this was an outcome of the solution for a specific objective. The average feed intake per pig for each diet within a feeding phase was predicted based on meeting the animal's requirements for growth. The net energy (NE) requirement for each feeding phase was defined in compliance with the NRC 2012 animal requirement tables (NRC, 2012a). Minimum nutrient levels in g/MJ of NE were then defined for each feeding phase, so that the digestible protein and macronutrient content of the feed would not be limiting for animal growth (NRC, 2012a). It was thus assumed that feed intake was driven by the animals need to meet its daily energy requirements; as such feed intake increased when diets of reduced energy density were fed (Kyriazakis and Emmans, 1995; Patience, 2012). The average predicted NE intake was constant for all diets. As all diets were nutritionally balanced the animals were expected to spend the same average number of days in the barn over the course of the G/F phase. When diets were formulated at reduced energy density, daily feed intake was expected to compensate for this. Any effects the increased daily intake may have had on gut fill were taken into account.

Average ingredient prices and availability in Ontario and Manitoba for 2015 were provided by Trouw Nutrition Agresearch, (derived from Statistics Canada data (Statistics-Canada, 2014b) - (see appendix H for the list of available ingredients and price ratios in each region). These were used to represent typical diet formulation scenarios for Eastern and Western Canada. Ontario and Manitoba produced around 24% and 23% of the total pigs marketed in Canada in 2011, respectively (Brisson, 2014). Importantly, corn was not considered as an available ingredient for the Western diets as is typical in many scenarios in this region; similarly, barley was not considered as an available ingredient in the Eastern diets (A. Pharazyn per comm, 2015).

The average gain: feed ratio over the G/F phase in the benchmark data for Canadian pigs was 0.365 with feed intake 264 kg per pig based on the mean start and finish weights. This was used as a starting point for the assumptions on average feed intake in this study. A dietary specification was defined which represented an industry standard to ensure feed: LW gain ratio was minimised within reasonable commercial constraints. The specifications of this “typical” diet are found in Table 5.2 and it was assumed that this diet ensured an average gain: feed ratio 0.365. Lower limits were defined for the nutritional density of the diets for each feeding phase. These were set at 95% of the energy content of the typical industry diet in the first 2 feeding phases and 92.5% for the latter 2 feeding phases. These restrictions were to ensure feed intake would not be restricted by gut fill, which can be caused by diets of lower nutrient density which contain a larger proportion of bulky feed (Kyriazakis and Emmans, 1995). These minimum specifications of the G/F diet for each phase can also be found in Table 5.2. For each ingredient a maximum inclusion rate was defined for each feeding phase in order to account for any anti-nutritional properties or other negative impacts on animal performance due to variability. These limits were based on guidance for pig farmers provided by the Ontario Ministry of Agriculture, Food and Rural Affairs (OMAFRA) (OMAFRA, 2012a) as well as peer reviewed studies in the case of some important co-products (see appendix I for further detail on ingredient inclusion limits).

The retention of N in finished pigs was calculated using the principles of Wellock et al (Wellock et al., 2003) and was assumed to be 0.0256 BW ± 0.00128. Retention of P and K were calculated using an allometric relationship of body composition to BW (Symeou et al., 2014) and were assumed to be approximately 0.005 BW ± 0.00025 and 0.002 BW ± 0.0001 respectively. For K this assumption represents a linear approximation around slaughter weight of a curvilinear relationship (Rigolot et al., 2010). All N, P and K not retained by the finished pigs were assumed to be excreted in faeces or urine. The predicted levels of nutrient excretion were required as inputs to the manure model.

Table 5.2 The nutritional specifications of the "typical" grower/finisher diet for Canadian pig systems. The lower limits permitted in the diet formulation rules used in this study are also shown.

Resource (g/kg unless otherwise stated)	Starter		Grower		Finisher		Late finisher	
	Typical	Lower Limit	Typical	Lower Limit	Typical	Lower Limit	Typical	Lower Limit
Net Energy (MJ/kg)	10.21	9.70	9.89	9.40	9.72	8.99	9.65	8.93
Dig Crude Protein	156.3	148.5	140.5	133.5	122.9	113.7	110.1	101.8
Dig Arg	10.5	10.0	8.8	8.3	7.2	6.7	6.3	5.8
Dig His	4.7	4.4	4.1	3.9	3.5	3.2	3.1	2.9
Dig Ile	6.1	5.8	5.3	5.1	4.6	4.3	4.0	3.7
Dig Leu	12.8	12.1	12.1	11.5	11.4	10.5	10.4	9.6
Dig Lys	10.4	9.9	9.2	8.7	7.3	6.8	6.5	6.0
Dig Met	3.2	3.0	2.7	2.6	2.5	2.3	2.2	2.0
Dig Phe	7.2	6.8	6.4	6.1	5.7	5.3	5.1	4.7
Dig Thr	6.3	6.0	5.8	5.5	4.9	4.5	4.4	4.1
Dig Trp	1.7	1.6	1.5	1.4	1.2	1.1	1.1	1.0
Dig Val	7.3	6.9	6.5	6.2	5.8	5.4	5.1	4.7
Dig Cys	2.7	2.6	2.7	2.6	2.5	2.3	2.3	2.1
Dig Meth + Cys	5.9	5.6	5.5	5.2	5.1	4.7	4.5	4.2
Ca	7.6	7.2	7.6	7.2	6.7	6.2	5.9	5.5
P	5.5	5.2	5.3	5.0	4.6	4.3	4.1	3.8
Dig P	3.1	2.9	2.8	2.7	2.3	2.1	1.9	1.8
K	6.6	6.3	6.2	5.9	5.6	5.2	5.0	4.6

5.3.8 Diets formulated

The process followed to formulate G/F diets for environmental impact objectives is shown as part of Figure 5.2. The average NRRU, AP, EP and GWP per kg of each ingredient as seen in Table 5.1 were added to the list of ingredient properties in the diet formulation tool. As well as this, equations which predicted the average environmental impact per kg of N, P and K excretion assuming an average mix of manure management practices were extracted from the manure sub-model of the LCA (described in detail in appendix D). This enabled the tool to account for the environmental impact resulting from predicted levels of nutrient excretion when formulating the diets. Thus for any diet formulated the average NRRU, AP, EP and GWP resulting from the feed supply chain and manure storage and application was predicted.

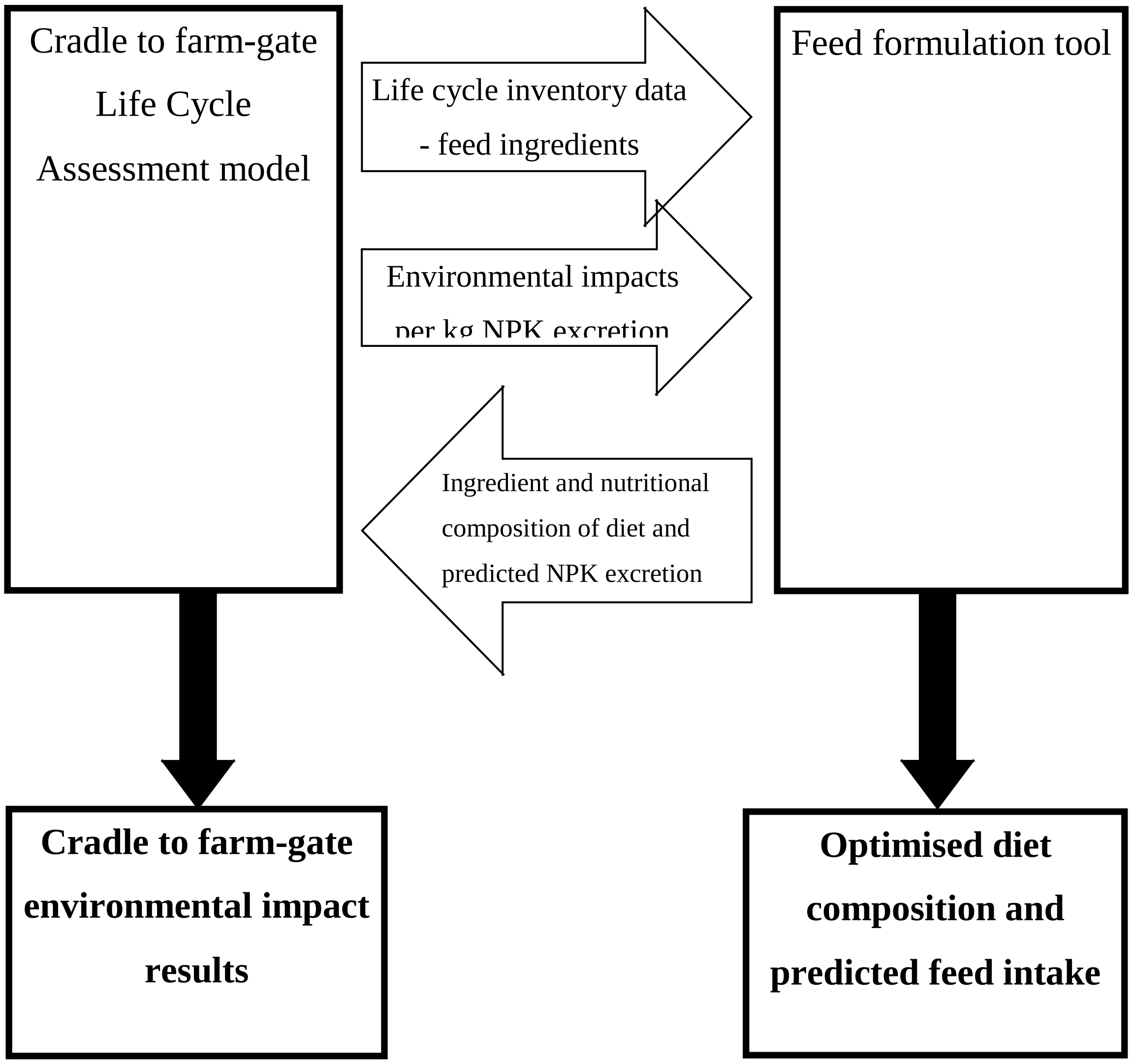


Figure 5.2 Schematic of the methodology followed in this study to formulate diets for environmental impact objectives.

The tool was used to formulate G/F diets for both economic and environmental impact objectives. Two diets were formulated for economic objectives: 1) to minimise feed cost per kg LW gain (least cost) and 2) to minimise feed cost per kg LW gain with a requirement to maintain a certain level of feed efficiency (least cost EFF). The NE content of the latter diet was fixed, so that feed: LW gain ratio was minimised within reasonable commercial constraints. The minimum specifications of this diet were the “industry standard” energy and nutrient levels shown in Table 5.2. This is a common commercial scenario, whereby diets are formulated for least cost without compromising feed efficiency (Ferguson, 2014). This diet was included to quantify whether this strategy has any benefit for the environmental impact of the system compared to considering feed cost alone.

Four diets were formulated to minimise the individual environmental impact categories NRRU, AP, EP and GWP. A further diet was formulated to minimise the combined environmental impact (least EI) of the G/F phase, as measured using the combined normalised levels of NRRU, AP, EP and GWP under the CML methodology (CML, 2002) with equal weighting. All diets formulated for environmental impact objectives were restricted to a 30% maximum cost increase in comparison to the least cost diet. Diets were formulated using linear programming for these objectives in both regional scenarios for ingredient prices and ingredient availability for Eastern and Western Canada. The resulting diets were optimal solutions based on the mean nutritional and environmental impact properties of the ingredients, as well as the mean impact levels associated with nutrient excretion calculated by the LCA.

5.3.9 Dietary comparisons in the LCA model

Accounting for the uncertainty in LCA is important to produce credible and reliable results (Lloyd and Ries, 2007). In this study an uncertainty analysis was used for statistical comparison of the diet formulations. The cradle to farm gate LCA model was hosted in the specialist software SimaPro 7.3.3®. All input parameters had a mean, associated distribution (e.g. normal, lognormal etc.) and standard deviation. The uncertainty in the environmental impact calculations was quantified using Monte-Carlo simulations. Variability in all characteristics of herd performance other than feed intake was assumed to be independent of feed composition in the G/F production stage. Feed intake for each simulation was a function of the energy density of the diet in relation to the average energy requirement of the herd over the G/F production stage. This requirement had a distribution to represent variation in feed intake due to genetic and environmental factors, which were assumed to be independent of the feed composition.

As shown in Figure 5.2 each diet was tested in the cradle to farm-gate LCA of pig farming systems in Eastern and Western Canada. In each case 1000 simulations of the model were run in order to calculate the NRRU, AP, EP and GWP of the system when adopting these diets. This number of simulations ensured the standard error of the mean (SEM) of the results for each impact category were low enough for good repeatability (see results of Chapter 3). Parallel Monte-Carlo simulations were used to compare all other diets to the least cost diet. The parallel simulations enabled the model to determine whether diets had resulted in any significant changes to the environmental impact levels of the system compared to the least cost scenario. This method of uncertainty analysis to distinguish between two scenarios in an LCA model was described in detail in Chapter 3. Briefly, uncertainties were categorised as either specific to the system (α) or shared between the systems being compared (β) (Leinonen et al., 2013, 2012). For each simulation a value for each parameter was randomly selected from the specified distribution input for this variable. Where parameters are shared between two scenarios being tested (for example corn yield (kg/hectare) when feeding two different diets containing corn), for each individual comparison the same point on the distribution is selected. In this case variation in all parameters, except the G/F diet composition, the resulting feed intake and nutrient excretion during the G/F phase and carcass yield were considered shared uncertainty in the comparisons. While the average energy requirement was variable to account for differences caused by animal and environmental factors, in each comparison the NE intake was the same for both diets. The key output of the simulations was the frequency in which the environmental impact of one scenario was greater or smaller than the second scenario for each impact category tested. Environmental impact levels were reported as significantly different in cases where $P < 0.05$ over 1000 parallel simulations of the LCA model. This allowed the model to account for shared uncertainty between two systems (in this case diets) modelled in the LCA, but provide a useful answer as to which diet is likely to cause greater environmental impact.

5.4 Results

5.4.1 Diet composition

The overall ingredient and nutritional composition of the diets formulated for Eastern and Western Canada are in Tables 5.3 and 5.4 respectively, along with the predicted feed cost and average feed intake per pig for each diet. For both regional scenarios the least cost diet had the lowest nutritional density and thus the highest average predicted feed intake over the G/F cycle of the diets formulated. The least cost EFF diet minimised feed intake (by design) and was cheaper per kg LW than all diets formulated for environmental impact objectives.

Of the diets formulated for environmental impact objectives the least NRRU diets was the most expensive in both regions, resulting in a 30% increase in feed cost in comparison to the least cost diet. The least GWP diets also resulted in large increases in feed cost per kg LW of 30% and 23% in the East and West Canadian scenarios, respectively. The least NRRU and least GWP diets raised feed costs significantly due to increased inclusions of relatively expensive protein meals (soybean meal and canola meal). The least AP increased feed costs by 12% in Eastern Canada and 16% in Western Canada compared to the least cost diet. The least EI diets were 12% more expensive than the least cost diet in both regions. The Least EP diet was the cheapest of the diets formulated for environmental impact objectives, increasing feed costs by 8% and 6% compared to the least cost diet in Eastern and Western Canada respectively.

In both regions the least GWP diet was the most energy dense of all the diets formulated for environmental impact objectives (along with the least EI diet in the east), with feed intake the same as the least cost EFF diet. The West Canadian least EI diet was less energy dense and thus average feed intake per pig was higher at 274 kg in comparison to 264 kg per pig in the Eastern scenario. The least EP diet reduced average feed intake by 3% in the Eastern scenario and 6% reduced average feed intake in the West. Compared to the least cost diet average feed intake was 5% lower for the least AP diet in the East Canadian scenario and 4% in the West. The least NRRU diets were the least nutritionally dense of the environmental impact objective diets with feed intake 2% lower in the east and 4% lower in the west in comparison to the least cost diet.

Table 5.3 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated for different objectives for Eastern Canada. All ingredient inclusion and nutrient levels shown are g/kg as fed unless otherwise stated. The average predicted feed intake and feed costs for each grower/finisher diet are also shown

Objective[1]	**Least cost**	**Least cost EFF**	**Least NRRU**	**Least AP**	**Least EP**	**Least GWP**	**Least EI**
Average feed cost (CAD/ kg live weight gain)	0.544	0.562	0.708	0.610	0.591	0.708	0.611
Average feed consumed (kg/pig)	280.5	264.0	275.8	265.4	272.5	264.0	264.0
Ingredient							
Canola Meal	42.77	51.05	100.00	95.69	96.39	0.00	71.18
Corn	574.99	706.29	232.13	443.17	580.35	237.67	480.57
Corn DDGS	36.79	0.00	0.00	113.88	53.10	0.00	0.00
Meat meal	0.00	0.00	39.83	0.00	0.00	40.99	0.00
Bakery Meal	0.00	0.00	94.01	94.08	94.24	94.05	94.05
Soybean meal	88.67	169.88	250.00	46.51	62.38	250.00	109.81
Wheat	0.00	0.00	0.00	0.00	25.63	0.00	0.00
Wheat Bran	0.00	0.00	0.00	50.00	50.00	50.00	50.00
Wheat shorts	231.29	45.11	261.53	86.64	0.00	260.60	136.25
Limestone	13.46	12.40	13.06	22.48	19.78	26.52	22.03
Dicalcium Phosphate	0.86	3.73	0.00	0.54	2.09	0.00	0.29
NaCl	4.22	4.77	2.41	3.22	2.92	3.19	3.41
Lysine HCL	2.35	0.86	0.00	3.70	3.35	0.00	2.18
DL Methionine	0.11	0.06	0.00	0.12	0.15	0.02	0.22
L Threonine	0.48	0.08	0.00	0.80	0.81	0.00	0.57
L Tryptophan	0.00	0.00	0.00	0.02	0.01	0.00	0.00
Soybean Oil	0.00	0.00	0.00	0.00	0.00	19.00	15.81
AV fat blend	0.00	1.49	2.96	34.91	4.68	13.70	9.37
Additives	4.01	4.26	4.08	4.24	4.13	4.26	4.26
Resource							
Net Energy (MJ/kg)	9.24	9.82	9.39	9.77	9.51	9.82	9.82
Dig CP	127.7	145.15	213.7	128.7	125.5	192.5	134.7
Dig Arg	8.6	10.1	16.3	7.8	7.6	14.8	9.2
Dig His	4.0	4.8	7.0	3.7	3.7	6.3	4.2

Objective[1]	Least cost	Least cost EFF	Least NRRU	Least AP	Least EP	Least GWP	Least EI
Dig Ile	5.0	6.0	9.0	4.8	4.7	8.1	5.3
Dig Lys	7.5	8.0	11.8	8.0	7.8	10.5	8.0
Dig Phe	6.2	7.2	10.3	6.0	5.8	9.4	6.4
Dig Thr	4.9	5.2	7.6	5.1	5.0	6.7	5.2
Dig Trp	1.4	1.6	2.7	1.3	1.3	2.4	1.6
Dig Val	6.2	6.9	10.5	6.1	5.8	9.4	6.4
Dig Cys	2.4	2.6	3.6	2.5	2.5	3.1	2.6
Dig Meth + Cys	4.9	5.2	7.2	5.1	5.0	6.2	5.1
Ca	6.5	6.9	9.9	10.2	9.4	14.5	10.0
P	5.2	4.8	7.8	5.1	4.7	7.3	5.0
Dig P	2.7	2.5	4.4	2.6	2.3	4.2	2.4
K	7.0	6.7	10.5	6.5	5.8	9.8	6.9
Gross Energy (MJ/kg)	16.7	16.3	17.3	17.4	16.4	17.6	17.0
Crude protein	165.5	175.4	271.3	166.3	156.6	242.5	170.4
Ash	45.5	42.2	60.7	57.6	52.4	69.8	57.9

[1] Least Cost, least feed cost per kg live weight gain. Least cost EFF, least cost / kg Live weight gain while maximising feed efficiency within commercial constraints. NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential. GWP, Global Warming Potential. Least EI, least combined environmental impact score.

Table 5.4 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated for different objectives for Western Canada. All ingredient inclusion and nutrient levels shown are g/kg as fed unless otherwise stated. The average predicted feed intake and feed costs for each grower/finisher diet are also shown

Objective[1]	**Least cost**	**Least cost EFF**	**Least NRRU**	**Least AP**	**Least EP**	**Least GWP**	**Least EI**
Average feed cost (CAD/ kg live weight gain)	0.536	0.550	0.690	0.623	0.567	0.656	0.599
Average feed consumed (kg/pig)	283.1	264.0	271.4	272.2	266.4	264.4	274.4
Ingredient							
Barley	0.00	0.00	0.00	579.38	0.00	353.32	489.80
Canola Meal	38.61	52.00	77.97	3.05	0.00	61.03	0.00
Corn DDGS	83.09	112.34	0.00	179.26	145.46	0.00	164.05
Meat meal	0.00	0.00	1.01	0.00	0.00	0.27	0.00
Field Peas	100.00	100.00	100.00	13.81	0.00	0.00	12.05
Soybean meal	5.40	13.95	250.00	59.35	34.67	250.00	57.03
Wheat	553.94	606.49	279.81	0.00	518.81	0.00	0.00
Wheat Bran	0.00	0.00	0.00	42.80	0.00	0.00	0.00
Wheat shorts	177.93	48.82	261.53	37.67	209.32	260.49	190.87
Limestone	12.59	11.67	11.21	21.75	25.52	22.03	24.12
Dicalcium Phosphate	2.71	6.11	0.00	0.55	2.40	0.00	0.21
NaCl	3.97	4.14	4.54	3.65	4.10	4.69	3.57
Lysine HCL	2.94	3.44	0.00	3.60	4.50	0.00	3.35
DL Methionine	0.05	0.08	0.00	0.19	0.13	0.00	0.15
L Threonine	0.47	0.66	0.00	0.77	0.88	0.00	0.68
L Tryptophan	0.00	0.00	0.00	0.04	0.00	0.00	0.01
Soybean Oil	0.00	0.00	0.00	7.80	14.55	0.00	20.00
AV fat blend	14.30	36.07	9.82	42.20	35.45	43.91	30.00
Additives	4.00	4.26	4.10	4.14	4.23	4.26	4.10

Objective[1]	Least cost	Least cost EFF	Least NRRU	Least AP	Least EP	Least GWP	Least EI
Resource							
Net Energy (MJ/kg)	9.20	9.82	9.55	9.52	9.79	9.80	9.45
Dig Arg	8.5	8.4	18.1	7.6	7.8	14.7	8.0
Dig His	3.7	3.8	7.6	3.6	3.7	6.4	3.7
Dig Ile	5.2	5.4	10.0	4.8	5.1	8.2	4.8
Dig Leu	10.8	11.5	17.6	11.1	11.4	14.4	11
Dig Lys	7.5	8.0	13.4	7.8	7.9	10.7	7.7
Dig Met	2.4	2.6	3.8	2.5	2.6	3.1	2.5
Dig Phe	6.7	7.0	11.5	6.5	6.8	9.7	6.5
Dig Thr	4.8	5.2	8.4	5.0	5.1	6.9	5.0
Dig Trp	1.5	1.6	3.0	1.3	1.5	2.5	1.3
Dig Val	6.4	6.5	11.4	6.2	6.4	9.5	6.3
Dig Cys	3.0	3.1	4.2	2.5	2.9	3.4	2.5
Dig Meth + Cys	5.4	5.7	8.1	5.0	5.5	6.5	4.9
Ca	6.4	6.9	6.7	9.5	11.2	10.6	10.3
P	5.6	5.7	7.4	4.8	5.6	6.2	5.2
Dig P	2.3	2.4	3.5	2.5	2.4	3.1	2.8
K	7.0	6.4	11.3	7.1	7.1	10.4	7.7
Gross Energy (MJ/kg)	16.5	16.7	17.3	17.6	17.3	17.8	17.8
Crude protein	179.3	180.9	297.7	170.1	178.1	246.3	174.4
Ash	45.2	43.4	53.2	51.6	57.3	64.3	56.7

[1] Least Cost, least feed cost per kg live weight gain. Least cost EFF, least cost / kg Live weight gain while maximising feed efficiency within commercial constraints. NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential. GWP, Global Warming Potential. Least EI, least combined environmental impact score.

The least cost EFF diet contained the largest amount of cereals (corn in the east and wheat/barley in the west) of all diets formulated. In both regions this diet contained the lowest levels of co-products (such as corn DDGS and wheat shorts), as well as an increased combined inclusion of oilseed meals (canola meal and soybean meal) compared to the least cost diet. All diets formulated for environmental impact objectives in Eastern Canada included the maximum allowed levels of bakery meal in the G/F diets. Similarly, with the exception of the least NRRU diet, all diets formulated for environmental impact objectives in

Eastern Canada contained the maximum amount of wheat bran. This was not the case for the wheat/barley based diets formulated in the Western Canada.

In both regions the least NRRU diet contained the lowest combined inclusion of whole cereals (wheat, barley and corn). The least NRRU diet contained no synthetic amino acid supplements or corn DDGS in either region. In both regions the least GWP and least NRRU diets were very similar: both contained high levels of wheat shorts and, in the East, bakery meal and meat meal. There was also an increased inclusion of soybean meal in the least GWP diets with very little synthetic amino acid supplementation compared to the least cost formulation.

5.4.2 Environmental Impacts – Eastern Canada

The environment impact results per kg of ECW from cradle to farm gate for the East Canadian diets when tested in the LCA model are in Table 5.5. The relative trade-offs of diets formulated for different objectives in terms of environmental impact, feed cost and feed intake are shown in Figure 5.3 for Eastern Canada. The least cost EFF diet reduced NRRU and GWP by 8% and 3%, respectively, compared to the least cost diet; levels of AP and EP were not significantly different between these two scenarios. The combined environmental impact score of the least cost EFF diet was marginally lower than the least cost diet by <1%.

Table 5.5 The environmental impacts per kg of Carcass Weight for grower/finisher diets in Eastern Canada formulated for different objectives.

Impact category[1]	Unit[2]	Least Cost	Least Cost EFF	Least NRRU	Least AP	Least EP	Least GWP	Least EI
NRRU	kg Sb eq	0.0063	0.0058	0.0033	0.0075	0.0071	0.0035	0.0054
AP	kg SO_2 eq	0.0548	0.0555^{NS}	0.0799	0.0520	0.0523	0.0688	0.0532
EP	kg PO_4 eq	0.0140	0.0140^{NS}	0.0208	0.0133	0.0132	0.0179	0.0135
GWP	kg CO_2 eq	2.09	2.03	1.80	2.14^{NS}	2.15	1.73	1.91
CML	<no units>	3.67E-13	3.65E-13	4.70E-13	3.62E-13	3.60E-13	4.13E-13	3.49E-13

[1] Least Cost, least feed cost per kg live weight gain. Least cost EFF, least cost / kg Live weight gain while maximising feed efficiency within commercial constraints. NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential. GWP, Global Warming Potential. Least EI, least combined environmental impact score. CML, CML methodology combined environmental impact score.

[2] eq, equivalent

^{NS} = Not significantly different from the Least Cost diet (P>0.05)

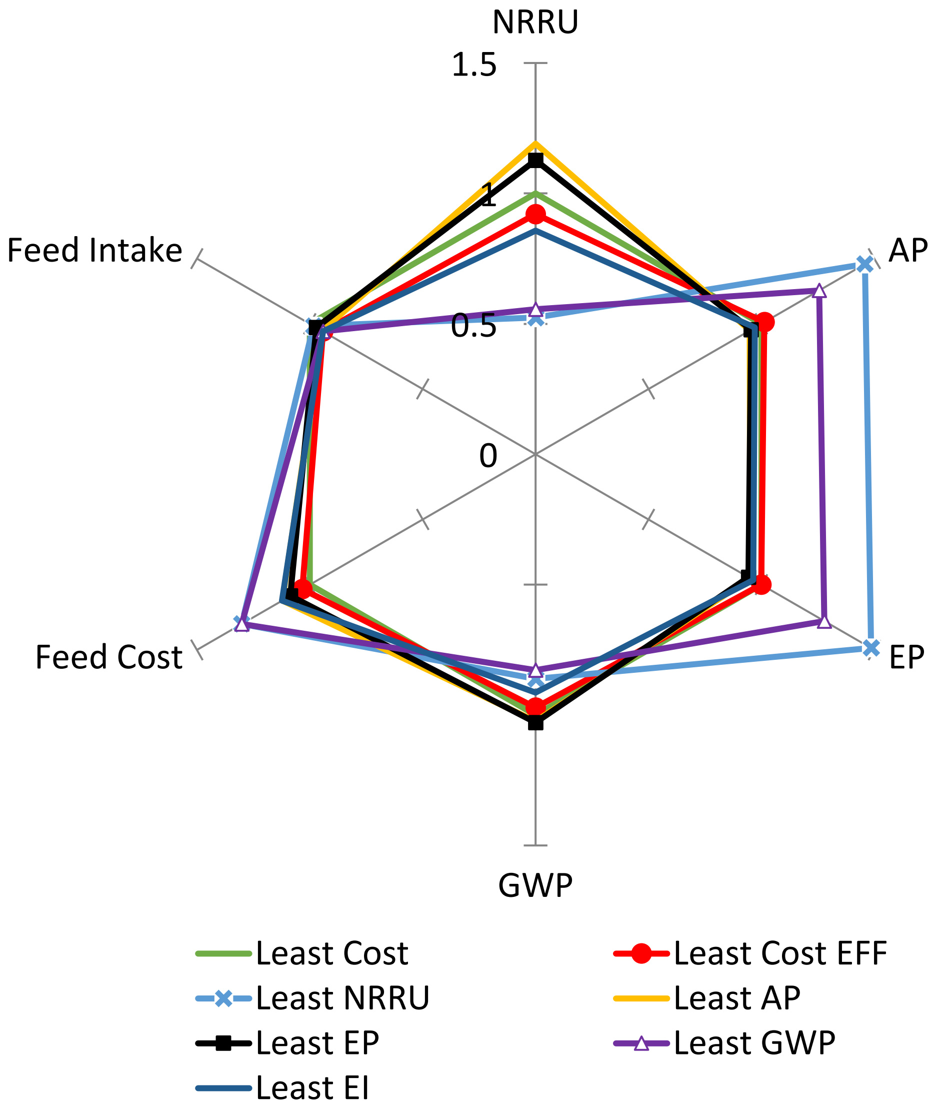


Figure 5.3 The environmental impacts, feed cost and feed intake per kg of Carcass Weight for grower/finisher diets in Eastern Canada formulated for different objectives, represented as a fraction of the results for the least cost diet. Least cost = least feed cost per kg live weight gain, Least cost EFF = least cost / kg Live weight gain while maximising feed efficiency within commercial constraints, NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. Least EI = least combined environmental impact score.

Reductions in NRRU (48%), AP (5%), EP (6%) and GWP (17%) were made when diets were formulated to minimise these impact categories in comparison to the least cost diet. The maximum reduction achieved in the combined environmental impact score was 5% when optimising the G/F diets for this objective compared to the least cost diet. In each case diets aimed at minimising the individual environmental impact categories resulted in increases in some of the other impact categories tested, compared to the least cost diet. The least NRRU diet also reduced GWP by 14%, but increased AP and EP by 46 and 49% respectively. Similarly the least GWP diet reduced NRRU 45% but increased AP by 26% and EP by 28%. The least AP diet increased NRRU by 19%, whilst EP was reduced by 5% with no significant difference in GWP. The least EP diet also meant that AP was 5% lower, however NRRU and GWP increased by 13 % and 3% respectively. The least EI diet did not increase any of the four environmental impact categories tested, the only diet formulated to achieve this.

5.4.3 Environmental Impacts - Western Canada

The environment impact results per kg of ECW from cradle to farm gate for the diets in Western Canada when tested in the LCA model are in Table 5.6. The relative trade-offs of diets formulated for different objectives in terms of environmental impact, feed cost and feed intake are shown in Figure 5.4 for Western Canada. The least cost EFF diet resulted in a 6% increase in NRRU and reduced AP by 4%, while EP and GWP did not change. The combined environmental impact score of the least cost EFF diet was 3% lower than the least cost diet.

Reductions in NRRU (46%), AP (17%), EP (10%) and GWP (24%) were made when diets were formulated to minimise these impact categories in comparison to the least cost diet. A 7% reduction was made in the combined environmental impact score per kg of ECW when this was the objective. Diets optimised to minimise the individual environmental impact categories resulted in increases in some of the other impact categories tested, compared to the least cost diet. The least NRRU diet also reduced GWP by 19% but increased AP and EP by 28%. Similarly the least GWP diet increased AP by 8%, EP by 16% with NRRU reduced by 37% compared to the least cost diet. The least AP diet increased NRRU by 28% and did not significantly alter EP or GWP. The least EP diet meant that AP was 1.5% lower, however NRRU increased by 12 % with no significant change in GWP. The least EI diet in the West reduced AP (17%), but did increase NRRU by (17%) with no significant difference in EP or GWP compared to the least cost diet.

Table 5.6 The environmental impacts per kg of Carcass Weight for grower/finisher diets in Western Canada formulated for different objectives.

Impact category[1]	Unit[2]	Least Cost	Least cost EFF	Least NRRU	Least AP	Least EP	Least GWP	Least EI
NRRU	kg Sb eq	0.00797	0.00848	0.00427	0.0102	0.0086	0.0050	0.0093
AP	kg SO_2 eq	0.0648	0.0624	0.0827	0.0535	0.0604	0.0703	0.0540
EP	kg PO_4 eq	0.0167	0.0160^{NS}	0.0214	0.0162^{NS}	0.0150	0.0193	0.0160^{NS}
GWP	kg CO_2 eq	2.31	2.33^{NS}	1.87	2.30^{NS}	2.23	1.75	2.21^{NS}
CML	<no units>	4.34E-13	4.22E-13	4.91E-13	4.09E-13	4.10E-13	4.38E-13	4.02E-13

[1] Least Cost, least feed cost per kg live weight gain. Least cost EFF, least cost / kg Live weight gain while maximising feed efficiency within commercial constraints. NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential. GWP, Global Warming Potential. Least EI, least combined environmental impact score. CML methodology combined environmental impact score.

[2] eq, equivalent

^{NS} = Not significantly different from the Least Cost diet (P>0.05)

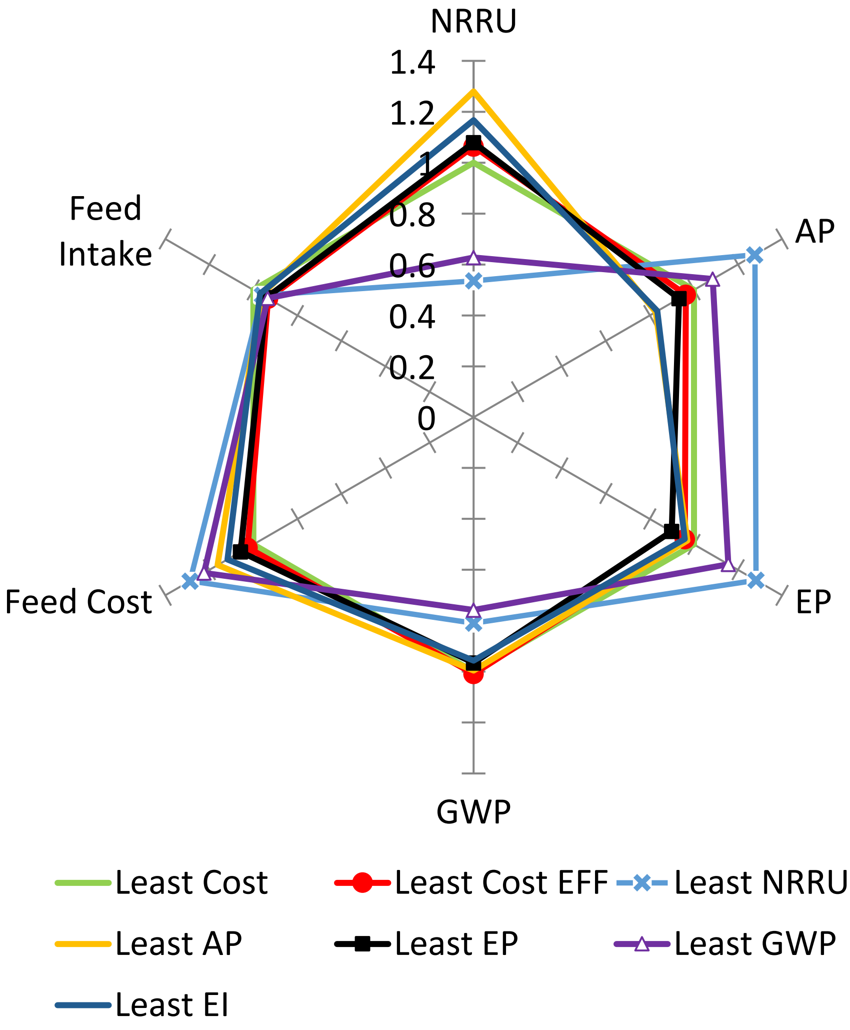


Figure 5.4 The environmental impacts, feed cost and feed intake per kg of Carcass Weight for grower/finisher diets in Western Canada formulated for different objectives, represented as a fraction of the results for the least cost diet. Least cost = least feed cost per kg live weight gain, Least cost EFF = least cost / kg Live weight gain while maximising feed efficiency within commercial constraints, NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. Least EI = least combined environmental impact score.

5.5 Discussion

As feed production and manure management are the main sources of environmental impact for pig production systems (Basset-Mens and Van Der Werf, 2005; Reckmann et al., 2013; Thoma et al., 2011), it is logical to consider diet formulation as a mechanism to reduce the environmental impact of pig production. In this study diets were formulated for the G/F production stage as this is where the majority of feed intake occurs per finished pig. There is also potential to formulate sow diets for environmental impact objectives to make reductions to the environmental impact of pig production systems. Although previous analysis of the farming systems modelled here showed that proportion of environmental impacts from this production phase is ~15% per kg ECW for most impact categories (see Chapter 3). Previous LCA studies have used scenario testing to demonstrate the potential for dietary changes to reduce the environmental impact of non-ruminant livestock systems (Eriksson et al., 2005; Garcia-Launay et al., 2014; Leinonen et al., 2013; Meul et al., 2012; Ogino et al., 2013). In this study we used a different approach by developing a novel methodology which integrated a cradle to farm-gate LCA model into a diet formulation tool to formulate diets for specific environmental impact objectives. Methodologies such as this one can allow nutritionists to integrate environmental impact objectives into diet formulations and for livestock producers to quantify the environmental impact of different feeding strategies. The methodology was associated with several challenges that are discussed below. The effectiveness of the methodology as a tool to reduce the environmental impacts of pig production systems and the strategies it identified to achieve this are then addressed.

5.5.1 Methodological Challenges

1) *Accounting for environmental impacts caused by ingredient choice, as well as nutrient excretion*

With the exception of Garcia-launay et al., (2015), previous LCA studies using life cycle inventory data to formulate diets which minimise the environmental impacts per kg of diet (Moe et al., 2014; Nguyen et al., 2012) have not taken into account the implications for nutrient excretion and the resulting environmental impacts. There are equations which can be integrated within animal growth models to predict nutrient excretion for a larger range of scenarios, using a more mechanistic approach than the one adopted in this paper (Ferguson, 2014; van Milgen et al., 2008). Previous studies have formulated diets where minimising nutrient excretion or levels of methane emissions were explicit objectives, as a way of incorporating environmental goals into least cost formulation (Moraes and Fadel, 2013; Pomar et al., 2007). These studies however, did not adopt a holistic LCA approach to quantify whether reductions in these specific emissions reduced the cradle to farm gate environmental impacts of the production system. The method developed in this paper accounted for the aggregated environmental impacts during manure management caused by N, P and K excretion when formulating diets for environmental impact objectives. It predicted the feed intake required for pigs to reach a target weight with any N, P and K not retained by the animal excreted in the urine or faeces. A component of the LCA of pig farming systems set out in Chapter 3 was integrated into the diet formulation algorithm to predict the NRRU, AP, EP and GWP which resulted from the storage and application to land of excreted nutrients as manure. This included an estimate of the potential of the nutrients contained in the manure produced to replace mineral fertilizers being applied to field in crop systems, an approach known as system expansion (Thomassen et al., 2008). This approach incorporates the potential benefits of replacing mineral fertilizers with manure as well as accounting for the extra emissions this may cause. To our knowledge this is the first time a diet formulation tool using a holistic LCA approach from cradle to farm gate has been developed to formulate livestock diets for environmental impact objectives.

2) *Formulating diets for multiple environmental impact objectives*

When formulating diets for environmental impact objectives in livestock systems, adopting a single metric is necessary in order to optimise diets for this purpose using linear programming. However, diets formulated to minimise one impact category may cause large increases in another type of environmental impact. If multiple environmental impact categories are to be accounted for when using linear programming a combined environmental impact score must be defined. Combining environmental impacts in a meaningful way is a significant methodological challenge to LCA practitioners; its subjective nature means there is little agreement on how best to approach it (Finnveden et al., 2009). In this study, the CML global normalisation methodology was adopted; there are many more complex methods for combining impacts which give various weightings to different types of impact (Goedkoop and Spriensma, 2001; Soares et al., 2006) but these methods are still based on subjective allocations of importance to the different impact categories. Such weightings are not currently recommended in the ISO standard for Life Cycle Impact Assessment (Finnveden et al., 2009; International Organisation for Standardisation, 2006b). It was not the purpose of this study to advance the discussion on how best to weigh environmental impacts. Any solution produced to minimise a metric for combined environmental impact is dependent on the methodology used to quantify it. Subjective choices such as which impact categories are included and how these categories are then weighted (to name only two) will hugely influence the outcome. The step of combining the impact categories provided the formulation tool with a framework to assess the trade-offs between decreases in one type of environmental impact and increases in another. Some methodologies have monetised the environmental impact categories using either the preferences of a panel, or the authors stated preferences to give a monetary value to different impact categories (Finnveden et al., 2006; Weidema, 2009). Further work to define acceptable methodologies for the monetisation of environmental impacts would enhance efforts to reduce the environmental impact of livestock systems. This could allow feed cost and environmental impacts to be integrated into a single objective to formulate diets which are economically and environmentally more sustainable.

3) *Allowing flexibility in the diet formulation rules to identify the optimal nutritional strategies for environmental impact objectives.*

Previous studies which have formulated diets for environmental impact objectives have done so for a fixed minimum nutritional specification for energy (MJ/kg) and nutrient content (g/kg) above which feed intake was assumed not to be affected (Moe et al., 2014; Nguyen et al., 2012). This is a fairly restrictive way to formulate diets and there is no consideration of the trade-off between environmental impact per kg of feed and feed intake. In this study the formulation algorithm accounted for the expected effect of energy density on feed intake and identified the optimum energy density across each feeding phase for a particular impact objective. This approach is common in commercial diet formulation as maximising gain to feed will not always result in the optimum outcome in terms of feed cost or other economic objectives (Ferguson, 2014). This was evident in the diets formulated to minimise feed cost per kg LW gain which were the least energy dense of all diets formulated in this study. Livestock diets have not been previously formulated for environmental impact objectives using this flexible approach to the nutritional density of the solution.

In this study improving gain: feed on a least cost basis reduced the environmental impacts of the farming system, as shown by the least cost EFF diet in both regions having a lower combined environmental impact score than the least cost diets. The diets formulated for least NRRU, AP and EP however, did not maximise gain: feed in both the East and West Canadian scenarios. The optimum energy density of the G/F diet was also different for each of the impact objectives. Similarly the least EI diets in the scenarios for Eastern and Western Canada also had differing energy densities, showing the need for flexibility when formulating diets for environmental impact objectives depending on the available ingredients. Formulating diets for a fixed minimum nutritional specification at an assumed feed intake would have restricted the ability of the tool to minimise both individual environmental impact categories, and the combined environmental impact score of the system. This is the first study to present a diet formulation algorithm which has the flexibility to identify the optimal nutritional density of livestock diets for different environmental impact objectives. The study demonstrated how environmental impact objectives can be integrated into modern diet formulation tools. The integration of diet formulation and LCA could be utilised to weigh the relative costs of reducing specific types of environmental impact from modern pig farms through diet manipulation. The approach could also be used to help modern pig production systems adapt and limit their liability to environmental taxes imposed on them.

5.5.2 Formulation strategies for environmental impact reduction

In most cases (with the exception of EP in the East and AP in the West) diets formulated for environmental impact objectives had a lower total inclusion of whole cereals (corn, wheat or barley) than diets formulated for economic objectives. This is because when formulating diets for environmental impact objectives, the environmental "cost" of production compared to the nutritional profile of these cereals is less favourable than their market value. When available, bakery meal was included at (or close to) maximum allowed levels in all diets formulated for environmental impact objectives. Bakery meal has relatively low levels of environmental impacts in the categories tested, and high nutritional value as an ingredient in diets fed to growing pigs (although there are concerns about its variability) (OMAFRA, 2012a). Apart from these two examples there were few uniform trends observed in the strategies adopted for different environmental impact objectives.

When minimising NRRU and GWP, high protein diets were formulated with increased inclusions of soybean meal and co-products such as wheat shorts, wheat bran and meat meal. Amino acid supplementation was not utilised when minimising NRRU and GWP. This contrasted with previous studies, conducted mainly in Europe, suggesting low protein diets with amino acid supplementation as a method of reducing GWP in pig production systems (Garcia-Launay et al., 2014; Mosnier et al., 2011a; Ogino et al., 2013). The reason for the difference is the majority of soybean meal used in European animal feed is imported from South America (Krautgartner et al., 2013) and is associated with recent land use change which carries a significant environmental impact penalty. Similarly, corn DDGS was also excluded from the least GWP and least NRRU diets because its production is associated with high levels of these impact categories (Table 5.1), due to energy inputs for drying and processing (Swiss Centre for Life Cycle Inventories, 2007). Previous LCA studies have also found that including corn DDGS in pig diets increased GWP in pig farming systems (Stone et al., 2012; Thoma et al., 2011).

In order to minimise AP and EP, diets were formulated with increased amino acid supplementation to minimise crude protein content. Other studies which have used scenario testing to assess the effect of amino acid supplementation in pig diets on the system's environmental impacts make similar conclusions (Garcia-Launay et al., 2014; Ogino et al., 2013). The results from both regions showed that increased inclusions of corn DDGS can be used as part of balanced G/F diets to minimise EP and AP in pig farming systems. This finding contradicts those of Chapter 4, which individually tested the effect of including DDGS in Canadian pig diets. The reason for the contradiction is due to differences in formulation objectives, with previous studies formulating for least cost rather than formulating for environmental impact objectives. This highlights the advantage of explicitly formulating pig diets for environmental impact objectives. A diet formulation algorithm can be used to formulate a balanced solution that includes ingredients which reduce the overall levels of a particular impact category, while simultaneously accounting for the trade-off between changes in feed intake and potential reductions in the environmental impacts per kg of the diet fed.

5.5.3 Effectiveness of optimisation as a strategy to reduce environmental impact in pig systems

The results of this study showed that through optimising G/F diets specifically for the purpose of reducing the environmental impact of pig production, it is possible to reduce the overall levels of NRRU, AP, EP and GWP in both corn and wheat/barley based diets. Relatively large proportional reductions were shown to be possible in the levels of NRRU and GWP in both regions when optimising to minimise the impacts individually. However, due to increases in EP and AP these diets increased the combined environmental impact score of the system. Such outcomes can only be considered a reduction in the environmental impact of the system if environmental impact categories other than the objective (e.g. GWP) are considered unimportant. This is difficult to justify in the case of pig farming systems which have been shown to cause relatively small levels of GWP compared to meat produced from ruminants (de Vries & de Boer 2010; Williams et al. 2006; Eshel et al. 2014). The results show the importance of considering multiple impact categories when using linear programming to optimise diets to reduce the environmental impacts of livestock systems.

Optimising G/F diets to minimise the combined environmental impact score resulted in relatively modest reductions (~5%) for the pig farming system in both regions. Cost was not the limiting factor for further reduction of the combined environmental impact score of the system; as the least EI diets in both regions were below the 30% increase limit on feed cost.

Further reductions in the combined environmental impact score through diet optimisation were restricted by the contrasting formulation strategies required to minimise NRRU and GWP compared to those for AP and EP. The solutions for least NRRU and least GWP were high protein diets which included large amounts of low value co-products; whereas the diets for least EP and AP, minimised dietary protein content and increased levels of amino acid and mineral supplementation. However, production of amino acid and mineral supplements ingredients had high associated NRRU and GWP. This meant the possible reductions in the combined environmental impact score were much lower than those for individual environmental impact categories such as GWP or NRRU.

There are examples of policies using financial penalties or rewards to provide economic incentives for livestock producers to reduce their environmental impacts. These have included taxes on spreading fertilizers in the EU (ECOTEC Research and Consulting, 2001) and payments to farmers for reducing the greenhouse gas emissions caused by farming activities in Australia (the carbon farming initiative) (Department of Environment - Australian Government, 2012). Methodologies like the one presented here, could be used to evaluate how livestock producers might adapt formulation strategies under such mechanisms, and whether these changes would reduce the cradle to farm gate environmental impact of livestock systems for a particular impact category. It is also possible to carry out sensitivity analyses in order to estimate the necessary levels of penalty or payments to incentivise changes which reduce the levels from cradle to farm gate by x% for a given impact category.

5.6 Conclusions

A modified diet formulation algorithm was designed which integrated important elements of an existing LCA model into a linear programme for diet formulation, in order to formulate G/F diets for environmental impact objectives. The flexibility of this approach allowed it to identify the optimum nutritional composition of the diets for a particular environmental impact objective as well as altering the ingredient composition. The optimum energy density of the G/F diet was different for each of the environmental impact objectives. Through optimising diets for individual environmental impact categories relatively large reductions in NRRU and GWP were found to be possible compared to the least cost diet, however these came at the expense of increases in AP and EP. The results showed that the easy solution to minimise environmental impacts is not always to feed a low energy by-product based diet. This was demonstrated by the least GWP diets, which in both regions were the most energy dense along with the least cost EFF diets. Diets were also formulated to minimise a combined environmental impact score for NRRU, AP, EP and GWP which enabled reductions in the environmental impacts of the system without any large increases in individual impact categories. Further work to define acceptable methodologies to combine and monetise different categories of environmental impact could allow feed cost and environmental impacts to be integrated into a single objective. This would allow nutritionists to formulate diets which are economically and environmentally more sustainable. This study demonstrated how environmental impact objectives can be integrated into modern diet formulation tools for livestock production systems using LCA.

Chapter 6: The potential of taxes to reduce environmental impact of pig farming systems through altering the composition of pig diets

6.1 Abstract

Environmental taxes are a form of incentive regulation available to governments in order to drive reductions in environmental impact. The aims of this study were to: 1) quantify the potential effect of environmental taxes on the composition of pig diets and the implications for the environmental impacts. 2) Examine the relationship between the level of tax and its effectiveness in reducing environmental impacts. Three taxes were investigated: a carbon tax on the feed ingredients as purchased, and two financial penalties on the spreading to fields of a kg of N and P in manure respectively. Each tax was integrated into a diet formulation algorithm for pig diets in Eastern and Western Canada and tested at a range of tax levels. The two regions use different feed ingredients and constitute a test for the consequences of different diet formulations. In each case diets were formulated to minimise feed cost per kg of live weight gain and the effect of the tax on feed cost as well as on predicted N and P excretion by the pigs were calculated. The results were then tested in a Life Cycle Assessment model representative of pig farming systems in the two regions, which calculated the potential effect of the diets on the aggregated environmental impacts of each farming system. The environmental impact implications of each environmental tax were quantified using four impact categories: global warming potential (GWP), acidification potential, eutrophication potential and non-renewable resource use. As the environmental tax levels increased, trigger points in the tax range caused dietary change which reduced levels of the targeted emission type i.e. GWP for the carbon tax, N excretion for the N tax and P excretion for the P tax. For all scenarios (except the P tax in the West) the largest reductions in the target emission per C$ increase in cost were achieved at the lower end of the tax range tested. The taxes on spreading N and P in manure did not significantly reduce levels of any environmental impact category tested in almost all cases. In many of the scenarios the environmental taxes altered the diet in a way which significantly increased levels of at least one of the environmental impact categories. These results have implications for the design of environmental taxes; they show the potential for taxes which target specific emissions, to increase system level environmental impacts in livestock production. They also demonstrate how system level environmental impact models can be used to quantify the cost effectiveness of a tax in reducing overall levels of environmental as well as the specific emission it targets.

6.2 Introduction

Pigouvian taxes are a form of incentive regulation available to governments in order to drive reductions in environmental impact (referred to in this context as environmental taxes). In comparison to more complex policy instruments, such as cap and trade, environmental taxes are relatively simple and give greater certainty regarding the monetary cost of the polluting emissions (Barthold, 1994). Pigou and other economists have long argued that environmental taxes are effective in forcing companies to internalise external costs related to their activities and ensure consumers are confronted with prices which reflect the full marginal social cost of a product (Hackett, 2011). Environmental taxes have often been used to incentivise environmental impact reduction in the agriculture sector; for example, some countries have introduced taxes on spreading Nitrogen and Phosphorus, which affect farm level decision making within livestock production systems (ECOTEC Research and Consulting, 2001; Sjöberg, 2005; Soil Service of Belgium, 2005). More recently due to concerns about climate change, there have been many proposals to introduce carbon consumption taxes as a mechanism to curb the carbon footprint of developed economies (World Bank, 2013).

When introducing environmental taxes to reduce the environmental impact of livestock production systems policy makers need to consider the following issues:

1) Which type/s of environmental impact is the tax designed to reduce? There are a number of environmental impact issues which are of concern regarding livestock production. While most attention has been given to the contribution of the livestock sector to greenhouse gas emissions (GHGs), other important environmental impact issues for the sector include the amount of crops grown for animal feed, water use and the contribution of nutrients excreted in animal manure to problems such as eutrophication and acidification (Bouwman et al., 2013; Eshel et al., 2014; Steinfeld et al., 2006). In many cases there may be more than one important environmental impact issue policy makers are trying to address regarding livestock production; it is therefore important that any taxes levied to reduce one type of environmental impact do not promote behaviour which increases other types of environmental impacts.

2) At which point in the production system should taxes be levied in order to be most effective? Firstly, this will depend on the environmental issue which is being targeted as different parts of the production system are most important for different types of impact. Generally, when considering the environmental impact of livestock production (and particularly for non-ruminant systems), the production of feed materials and the storage and

disposal of manure are the most important aspects of the production system for most impact categories (Leinonen et al. 2012; Williams et al. 2006; Basset-Mens & Van Der Werf 2005). Issues of practicality are also a factor in this decision, a tax must be levied on an aspect of the system which can be measured reliably in order to be practical. Preferably any tax should allow livestock producers to alter production practices to reduce levels of the type of pollution which are targeted by the tax and thus their liability.

3) At what penalty level should any environmental tax be set? Environmental taxes usually aim to reduce behaviour which is harmful to the environment rather than to raise large amounts of extra revenue (Fullerton et al., 2010). In order to be socially acceptable environmental taxes should not unduly penalise domestic industries, thus making them vulnerable to cheap imports which do not have to adhere to the same regulations. In relation to climate change this phenomenon is commonly referred to as "carbon leakage" (European Commission, 2009). As such environmental taxes should be designed to reduce environmental impact in the most cost effective manner possible.

In cases where environmental taxes are implemented on livestock systems, they can influence decision making within the sector, including the case of formulating animal diets. In order to quantify the implications of adopting different diets in livestock systems for environmental impacts these must be quantitatively modelled. Life Cycle Assessment (LCA) is a generally accepted method to evaluate holistically the environmental impact during the entire life cycle of a product or system (Guinée et al., 2002). Recently, researchers have used LCA modelling to present methodologies which integrate environmental impact considerations into diet formulation models, in order to formulate diets which restrict or minimise the environmental impact of livestock production (Moe et al., 2014; Nguyen et al., 2012). Chapter 5 showed when attempting to reduce the environmental impact of pig farming system through diet formulation, when diets were optimised to minimise a single environmental impact category, large increases in other types of environmental impact maybe caused.

Here we use a diet formulation tool designed for pig farming systems in Canada, combined with an LCA model of these systems as a case in point to investigate the potential implications of environmental taxes on diet formulation and the environmental impact of livestock systems. Pig diets in Eastern Canada are typically based on maize similar to USA pig diets (Thoma et al., 2011), whereas pig diets in Western Canada use wheat and barley as the main cereal component/s (Patience et al., 1995), as would be common for European pig diets.

The aims of this study were to:

1) quantify the potential effect of environmental taxes on the composition of pig diets, and the implications for the environmental impacts of the production system, assessed using multiple environmental impact categories
2) examine the relationship between the level of tax and its effectiveness in reducing environmental impacts through modelling each tax scenario at incremental levels of financial penalty.

Three taxes were each tested in a novel diet formulation algorithm: a carbon tax on the ingredients as purchased for feed and two financial penalties on the spreading (per kg) of N and P in manure respectively. It was expected that the diet formulation algorithm would respond to these taxes and alter the diets to meet their respective objectives; namely reducing the carbon footprint of the diet and reducing N and P excretion. However, we hypothesised that this may have unintended consequences and there would be trade-offs with policies aimed at reducing one type of impact increasing other types of environmental impact caused by the farming system.

6.3 Materials and Methods

6.3.1 The system considered

Modern pig farming systems can be considered to have 3 distinct production phases; 1) gestation and farrowing - where piglets are produced by breeding sows, 2) the nursery or weaning phase when pigs are separated from their mother and 3) the grower/finisher (G/F) phase where pigs are fattened from around 30kg to slaughter weight (PorkCheckoff, 2009). Figure 6.1 shows the major components of this system when considered in an LCA model; from the production of feed ingredients to animals shipped for slaughter at the farm gate. Benchmark data from 2012 on Canadian pig farms showed that 78% of feed consumed per pig produced and at least 75% of the environmental impacts occurred during the G/F phase (Chapter 3). This study therefore, concentrated on the potential effect of environmental taxes on diets formulated for the G/F phase of production only. The breeding and nursery production stages were treated as independent to the G/F phase in this study and remained constant for all comparisons made.

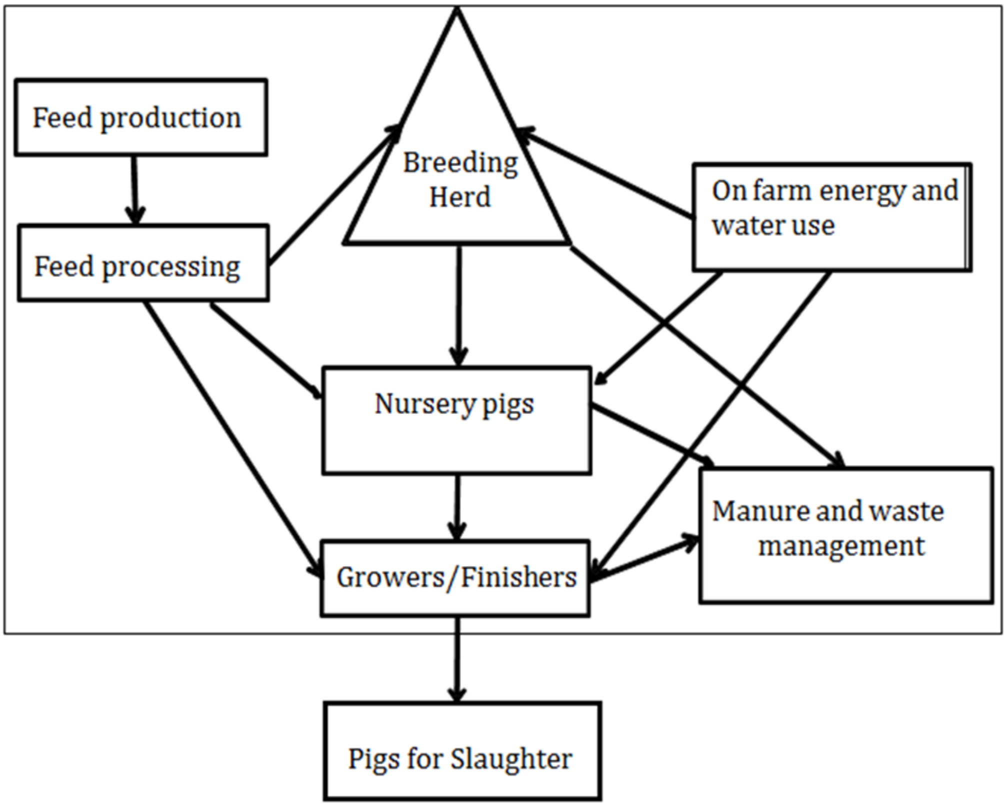


Figure 6.1 The structure and main components of the pig production systems as considered by the Life Cycle Assessment model.

6.2.2 Diet Formulation

A linear programming algorithm for diet formulation was used to formulate G/F diets for each taxation scenario; the diet formulation rules used are described in detail in Chapter 5. In this study all diets were formulated to minimise feed cost per kg live weight gain (least cost) for the G/F phase in each tax scenario. The predicted start weight of the pigs in the diet formulation algorithm was fixed a*t* 27.4 kg with a finish weight of 124 kg for the G/F phase, based on benchmark data collected for a previous LCA study of Canadian pig farming (Chapter 3). Minimum nutrient levels in g/MJ of Net Energy (NE) were defined for each feeding phase, so that the protein and macronutrient content of the feed would not be limiting for animal growth (NRC, 2012a); it was assumed that feed intake enabled animals to meet their energy requirements (Kyriazakis and Emmans, 1995; Patience, 2012). Thus the average predicted NE intake was constant for all diets. The main nutritional specifications of the "typical" Canadian diet are found in Table 6.1 and it was assumed that this diet ensured an average gain: feed ratio 0.365 kg/kg based on data collected for a previous LCA study of Canadian pig farming (see Chapter 3). Lower limits were defined for the energy density of the diets for each feeding phase to ensure feed intake would not be restricted by gut fill. This can be caused by diets of lower energy density which contain a larger proportion of bulky feed (Kyriazakis and Emmans, 1995). These minimum specifications of the G/F diet for each phase can also be found in Table 6.1

Table 6.1 The nutritional specifications of the “typical” grower/finisher diet for Canadian pig systems. The lower limits permitted in the diet formulation rules used in this study to ensure feed intake was not affected by issues such as gut fill are also shown.

	Starter		Grower		Finisher		Late finisher	
Resource (g/kg unless otherwise stated)	Typical	Lower Limit	Typical	Lower Limit	Typical	Lower Limit	Typical	Lower Limit
Net Energy (MJ/kg)	10.21	9.70	9.89	9.40	9.72	8.99	9.65	8.93
Digestible Crude Protein	156.3	148.5	140.5	133.5	122.9	113.7	110.1	101.8
Digestible Lysine	10.4	9.9	9.2	8.7	7.3	6.8	6.5	6.0
Digestible Methionine	3.2	3.0	2.7	2.6	2.5	2.3	2.2	2.0
Calcium	7.6	7.2	7.6	7.2	6.7	6.2	5.9	5.5
Phosphorus	5.5	5.2	5.3	5.0	4.6	4.3	4.1	3.8
Digestible Phosphorus	3.1	2.9	2.8	2.7	2.3	2.1	1.9	1.8
Potassium	6.6	6.3	6.2	5.9	5.6	5.2	5.0	4.6

There were 5 broad groups of ingredients used in the diet formulation algorithm; 1) whole cereals such as wheat and maize, 2) protein meals such as soybean meal and canola meal, 3) co-products of other production processes, such as wheat shorts from flour milling and corn dried distillers grains with solubles (DDGS), 4) supplements such as crystalline amino acids or minerals and 5) fats such as vegetable oil blends or rendered animal fat. Upper limits were placed on the inclusion of individual ingredients in the diets, so that issues of palatability or variability in specific ingredients did not adversely affect feed intake or animal growth (as described in Chapter 4). These were based on advice on diet formulation for pigs from the Ontario Ministry of Agriculture Food and Rural Affairs (OMAFRA, 2012a), the full list can be found in appendix I. Nutritional values for all ingredients in the diets were primarily taken from the Stein Monogastric Nutrition Laboratory ingredient matrix (Stein Monogastric

Nutrition Laboratory., 2014). Average ingredient prices and availability in Ontario and Manitoba for 2015 were provided by Trouw Nutrition Agresearch, derived from Statistics Canada data (Statistics-Canada, 2014b). The price ratios and available ingredients for Eastern and Western Canada can be found in the appendix H.

The diet formulation algorithm had two main features which enabled it to modify the diet in response to the environmental taxes tested: 1) Diets were not formulated for a fixed nutritional density, rather this was an outcome of the solution for the scenario tested. The average feed intake per pig for each diet within a feeding phase was predicted based on meeting the animal's requirements for growth. For the carbon tax this meant the model was able to weigh the trade-off between adapting the diet to reduce the carbon tax liability per kg of diet and any increases in feed intake caused by adapting the diet. 2) The excretion levels of key nutrients Nitrogen, Phosphorus and Potassium were predicted for each diet formulated. In the scenarios for taxes on spreading N and P contained in manure, the cost of spreading the predicted nutrient excretion was added to the feed cost and the combined cost was minimised as part of the diet optimisation. As such the model was able to strike a balance between the costs of feed ingredients against the costs incurred from nutrient excretion due to taxes on spreading manure.

6.3.3 Quantifying environmental impacts

The environmental impacts resulting from all diets formulated in this study were calculated using an LCA model of pig systems in Eastern and Western Canada (see Chapter 3 for full description). The system boundaries of the LCA were cradle to farm-gate and the functional unit was 1 kg expected carcass weight (ECW). There were three main compartments of material flow considered in the LCA model: 1) the production of feed ingredients, 2) the consumption of feed, energy and other materials for on-farm pig production and 3) the storage and land application of manure. Further details on the inventory data used to calculate the environmental impacts can be found elsewhere in the thesis and in the supplementary material; see Chapter 4 for details regarding feed ingredients, appendix C for data regarding farm energy use and appendix D for details of the manure model.

The environmental impacts of the system were quantified by the LCA using four environmental impact categories. Three of these categories quantified negative impacts resulting from emissions caused by the system; Acidification Potential (AP), Eutrophication Potential (EP) and Global Warming Potential (GWP). Reducing GWP caused by the production system would be the objective of a carbon tax. The impact categories AP and EP were considered as they quantify the main environmental impacts which result from the storage and spreading of animal manure. The aim of taxes on spreading N and P in manure as fertilizer is to reduce the systems contribution to these issues. A fourth impact category quantified the systems Non-Renewable Resource Use (NRRU) and was included because of the relatively high usage of cereals and oil seed meals in pig diets, which have a significant input of resources such as fertilizers (Steinfeld et al., 2006). When modelling a complex supply chain, as is the case for animal feed, the inputs to a process (wheat, water, energy etc.) are shared between the different multiple outputs (co-products) resulting from these processes, and the environmental impacts associated with them must be allocated. Economic allocation was used as the methodology for co-product allocation throughout the feed supply chain as advised in the FAO Livestock Environmental Assessment and Performance (LEAP) partnership recommendations (FAO, 2014a). The price ratios found in the appendix G were used for the purposes of economic allocation.

6.3.4 Uncertainty Analysis in the LCA model

In this study an uncertainty analysis was used for statistical comparison of the diet formulations. The cradle to farm gate LCA model was hosted in the specialist software SimaPro 7.3.3®. All input parameters had a mean, associated distribution (e.g. normal, lognormal etc.) and standard deviation. The uncertainty in the environmental impact calculations was quantified using Monte-Carlo simulations. Variability in all characteristics of herd performance other than feed intake was assumed to be independent of feed composition in the G/F production stage. Parallel Monte-Carlo simulations were used to compare all diets formulated at different tax levels to the no tax least cost diet. The parallel simulations enabled the model to determine whether diets had resulted in any significant changes to the environmental impact levels of the system compared to the least cost scenario. This method of uncertainty analysis to distinguish between two scenarios in an LCA model was described in detail in Chapter 3. The key output of the simulations was the frequency in which the environmental impact of one scenario was greater or smaller than the counterfactual scenario for each impact category tested. Environmental impact levels were reported as significantly different in cases were $P < 0.05$ over 1000 parallel simulations of the LCA model.

6.3.5 Taxation Levels

Diets were formulated for three different taxation scenarios; a carbon tax, a tax on spreading N contained in manure (N tax) and a tax on spreading P contained in manure (P tax). Each tax was tested at a variety of taxation levels on diets formulated in the two regions of Canada. In each case the output from the diet formulation algorithm was a diet composition which minimised feed cost/ kg LW gain during the G/F phase of pig production, as well as the predicted feed intake and feed cost for this diet. Each diet was then input into the LCA model described above in order to predict the environmental impacts of the system when adopting that diet as represented in the schematic shown in Figure 6.2.

The carbon tax was added to the price of each ingredient in the feed formulation algorithm based on the average GWP per kg of product for each ingredient. The tax was calculated using inventory data from the LCA model of Canadian pig systems (see Chapter 3), the GWP values used per kg of each ingredient in the feed formulation algorithm can be found in Table 6.2. The effect of a carbon tax on least cost G/F diet formulations was tested between 10-70 Canadian Dollars (C$) per tonne of CO_2 equivalent at increments of C$10. The levels tested reflected a range of valuations that governments and companies have placed on GHGs through carbon taxes and carbon shadow prices in an effort to tackle climate change (World Bank, 2013). Moreover many companies (including Google, Disney, Walmart and Exxon Mobil) are now using an internal carbon price as part of their business planning strategies, with those disclosed ranging from C$8-82 per tonne of CO_2 (CDP North America, 2013).

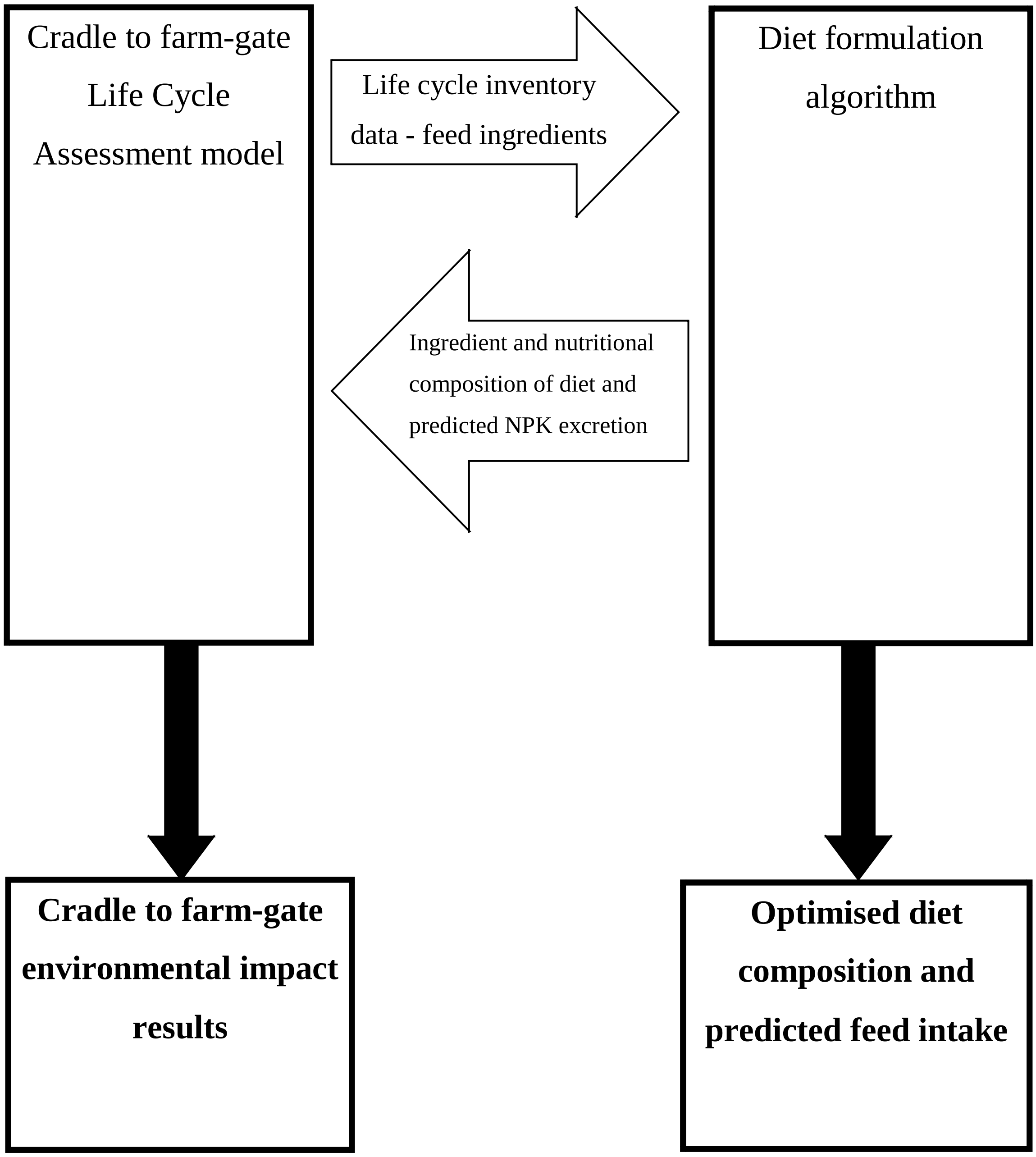


Figure 6.2 Schematic of the methodology followed to formulate diets in different environmental tax scenarios and then test these diets in the life cycle assessment model to determine the resulting environmental impacts.

Scenarios were modelled for taxes applied to excess N and P in the G/F diets which was excreted in manure and spread to field. The extra costs resulting from the tax were added to the overall feed cost within the formulation model and accounted for in the least cost formulation. Taxes on spreading N, P and K in fertilizer have been introduced at various levels in European countries such as Austria, Sweden and The Netherlands since the 1980's (ECOTEC Research and Consulting, 2001; Sjöberg, 2005; Soil Service of Belgium, 2005). The upper limit of tax levels tested was purposefully restricted to a maximum 25% increase in the overall cost of feed + manure spreading in either regional scenario. The effect of a tax on spreading Nitrogen contained in manure was tested in this study between C$0.5-3 per kg of N spread in manure on fields at increments of C$0.5. The effect of a tax on spreading Phosphorus contained in manure was tested between C$2.5-15 per kg of P spread in manure on fields at increments of C$2.5. The levels of N and P taxation tested in the formulation algorithm reflected a range of taxes found to have been implemented by governments across Europe of approximately C$0.1-3.6 per kg N and C$0.2-14.1 per kg of P_2O_5 contained in fertilizer spread to field depending on the conditions of the specific tax regime (ECOTEC Research and Consulting, 2001).

Table 6.2 Average environmental impacts per kg for all feed ingredients included in grower/finisher diets tested. Inventory data for these ingredients was compiled as part of a previous life cycle assessment studies of Canadian pig farming systems (see Chapters 3 & 4).

Impact category [1] **Unit** [2]	**NRRU** **kg Sb eq**	**AP** **kg SO2 eq**	**EP** **kg PO4 eq**	**GWP** **kg CO2 eq**
Barley	2.18E-03	5.36E-03	2.69E-03	0.38
Canola meal	1.39E-03	7.97E-03	1.59E-03	0.30
Canola oil	3.84E-03	2.20E-02	4.40E-03	0.84
Maize	1.71E-03	5.13E-03	1.11E-03	0.39
Soybean meal	5.70E-04	4.11E-03	8.71E-04	0.15
Wheat	1.84E-03	1.01E-02	2.04E-03	0.43
Meat (pork) meal	1.05E-03	2.46E-04	6.16E-05	0.13
Corn DDGS	6.51E-03	1.13E-03	2.66E-04	0.78
Wheat Bran	1.02E-03	5.56E-03	1.12E-03	0.24
Wheat shorts	5.12E-04	2.78E-03	5.59E-04	0.12
Field Peas	1.32E-03	2.31E-03	2.72E-03	0.58
Bakery meal	5.17E-04	1.41E-03	2.60E-04	0.08
Animal-vegetable fat blend	2.57E-03	1.01E-02	2.06E-03	0.49
Soybean oil	1.51E-03	1.09E-02	2.30E-03	0.40
HCL-Lysine	3.51E-02	2.12E-02	9.97E-03	4.81
L-Threonine	3.51E-02	2.12E-02	9.97E-03	4.81
FU-Methionine	3.64E-02	7.54E-03	1.70E-03	2.95
L-Tryptophan	7.01E-02	4.24E-02	1.99E-02	9.62
Sodium Chloride	1.21E-03	8.97E-04	6.68E-04	0.18
Dicalcium Phosphate	9.40E-03	2.68E-02	3.63E-04	1.51
Limestone	1.31E-04	1.03E-04	3.58E-05	0.02

1 NRRU, Non-renewable resource use. AP, Acidification Potential. EP, Eutrophication Potential, GWP, Global Warming Potential.

2 eq, equivalent

6.4 Results and discussion

6.4.1 Carbon tax

The ingredient compositions of the diets formulated at different levels of carbon taxation are shown in Figures 6.3a and 6.3b. The relative feed cost, feed intake, N excreted, P excreted, NRRU, AP, EP and GWP for each tax level are shown in Figures 6.4a and 6.4b for the Eastern and Western Canadian scenarios respectively, as a ratio compared to the no tax diet. In both regions the carbon tax produced reductions in the overall GWP caused by the farming system at all levels of taxation tested ($P<0.05$).

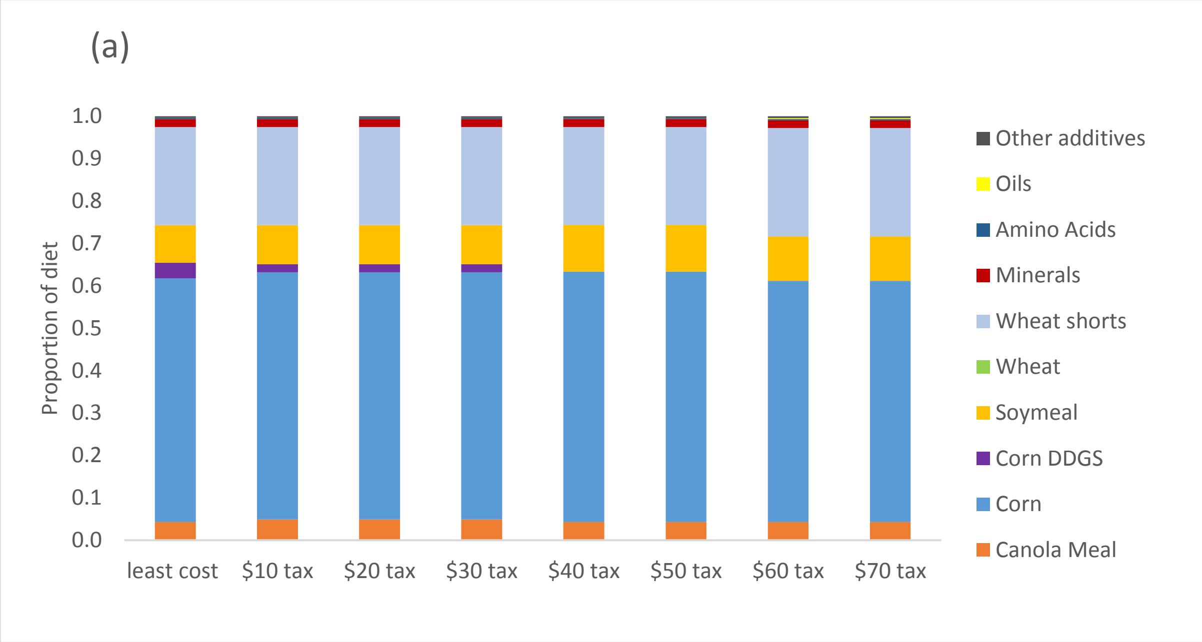
In the East Canadian scenario, all diets for tax levels of C$40 per tonne CO_2 eq and above reduced GWP by 4% ($P<0.01$) compared to the no tax scenario. At C$40 per tonne CO_2 eq the feed cost increased by 5%. At tax levels above C$40 per tonne, there were further changes to the ingredient composition of the diets and increases in cost, but there was little further reduction in levels of GWP. The carbon tax also reduced NRRU at all tax levels tested in the East Canadian scenario ($P<0.001$); at C$40 per tonne CO_2 eq and above NRRU was reduced by 11%. There was no significant difference in AP or EP caused by the system for any of the diets formulated under a carbon tax compared to the no tax scenario. Predicted N excretion remained constant as carbon tax increased in the East while P excretion was marginally reduced.

As the carbon tax levels increased, two trends were observed in terms of ingredient composition for the East Canadian scenario which reduced GWP and NRRU. Firstly, all levels of carbon tax caused a decrease in the amount of corn DDGS included in the diet compared to the no tax scenario, as corn DDGS had high levels of GWP per kg associated with it compared to other ingredients accordingly (see Table 6.2). Secondly at tax levels of C$40 per tonne CO_2 eq and above, soybean meal inclusion in the diet was greater than 100g/kg in the diet compared to 88g/kg in the no tax scenario. This meant a slightly lower inclusion of synthetic amino acids in the diet; production of these is also associated with high levels of GWP (Table 6.2). The nutritional density of the diets increased marginally at carbon tax levels above C$40 per tonne CO_2 eq in the East with average predicted feed intake 1% reduced compared to the no tax scenario.

In the West, a maximum reduction of 9% in GWP was observed compared to the no tax scenario at taxes of C$60 per tonne CO_2 eq and above ($P<0.001$), increasing feed cost by 10%. A carbon tax of C$40 per tonne CO_2 eq reduced GWP by 8% at 7% cost increase compared to the no tax scenario ($P<0.001$). All levels of carbon tax also reduced NRRU

($P<0.001$), with tax levels of C$40 per tonne of CO_2 equivalent and above causing at least a 19% reduction compared the no tax scenario. In the West taxation levels of C$40 per tonne of CO_2 equivalent and above caused increases in AP and EP of between 2-3% for both categories ($P<0.01$). However, in all scenarios tested the increases in AP and EP were smaller than the reduction in GWP as a percentage of impact levels in the no taxation scenario. Predicted N excretion increased by as much as 8% and P excretion by up to 4% at the higher levels of carbon tax in the West which in part explained the increases in AP and EP.

Similar to the Eastern scenario as levels of carbon tax were increased the least cost diet included less corn DDGS due to high levels of tax on this ingredient. The amount of soybean meal included in the G/F diets increased with carbon tax levels driving a reduction in the use of amino acid supplements which were also subject to high levels of tax. The inclusion of wheat shorts in the diet increased from 180 g/kg in the no tax scenario to a maximum of 260 g/kg, as wheat shorts had relatively low GWP and thus tax liability per kg (Table 6.2). The nutritional density of the least cost G/F diets did not change at any level of carbon tax tested.



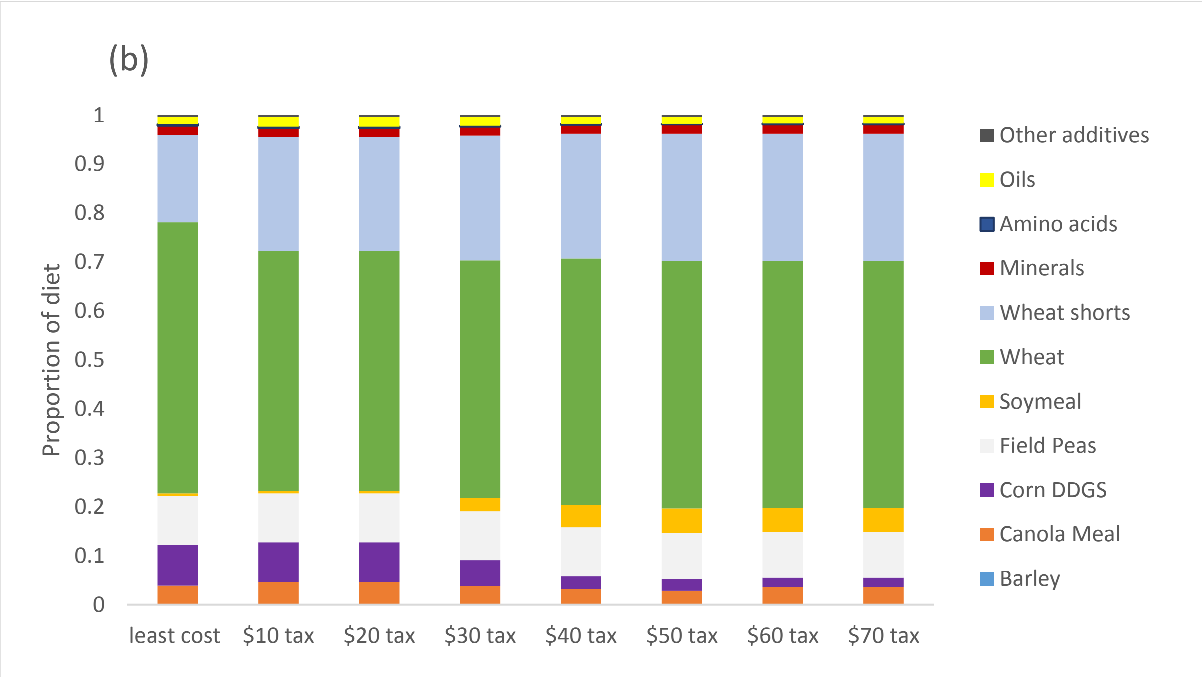


Figure 6.3 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent. C$ = Canadian Dollars

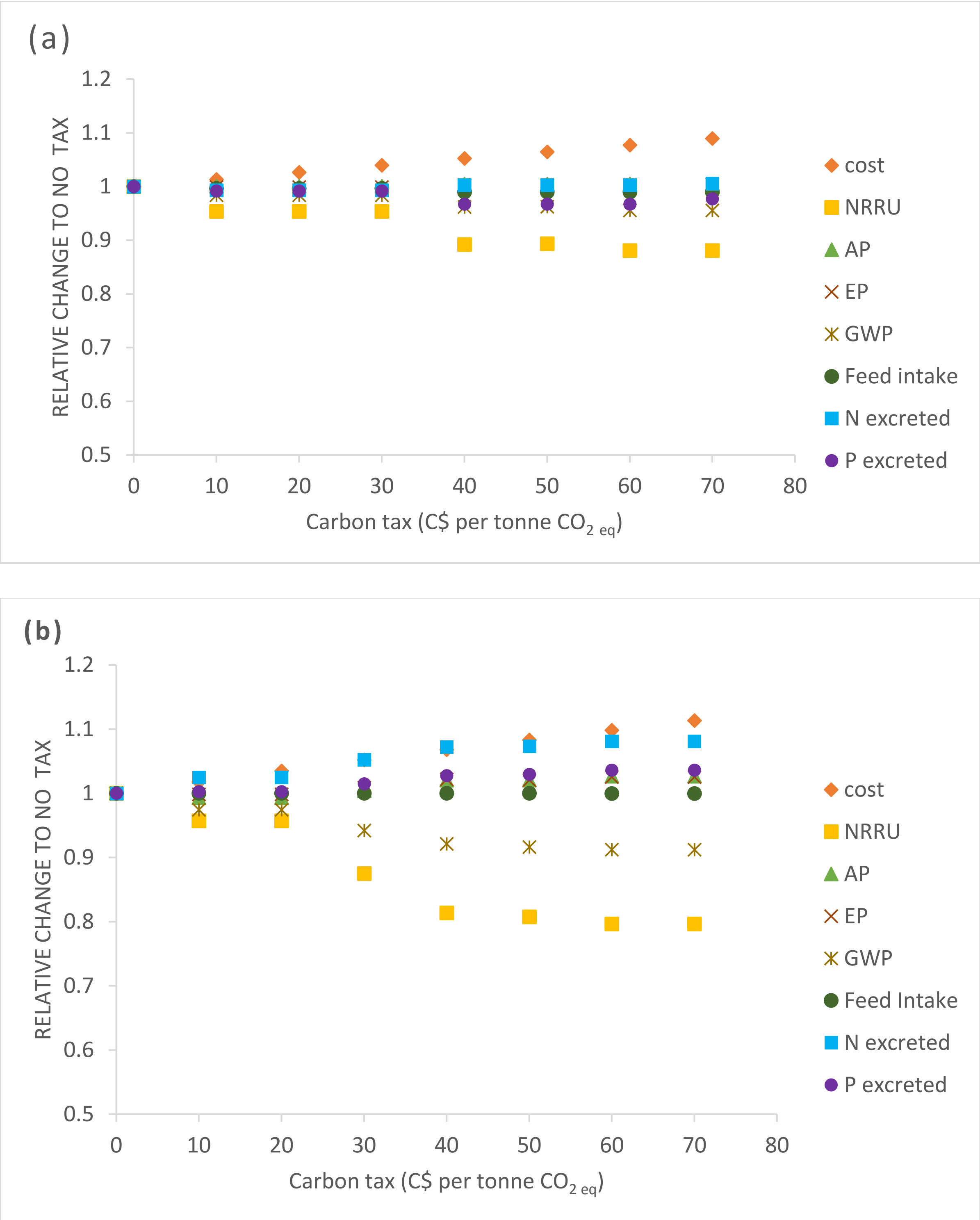


Figure 6.4 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq). NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. C$ = Canadian Dollars

Figure 6.5 shows the relative reduction in GWP per C$ cost increase for the different tax levels tested in the scenarios for Eastern and Western Canada. In both cases the lowest level of taxation tested produced the largest relative reduction in GWP per C$ increase in feed cost. The graphs show there were a couple of trigger points whereby increasing the level of Carbon tax caused changes in to the diet formulation produced greater reductions in GWP per C$ cost increase than the previous level tested. These were at C$40 in the East Canadian scenario as well as at C$30 and C$40 in Western Canada. Despite this, the general trend in both scenarios was for a diminishing relative reduction in GWP per C$ cost increase caused by the carbon tax as tax levels were increased. At all tax levels the reduction in GWP per C$ cost increase was greater for the West Canadian scenario compared to the East Canadian scenario.

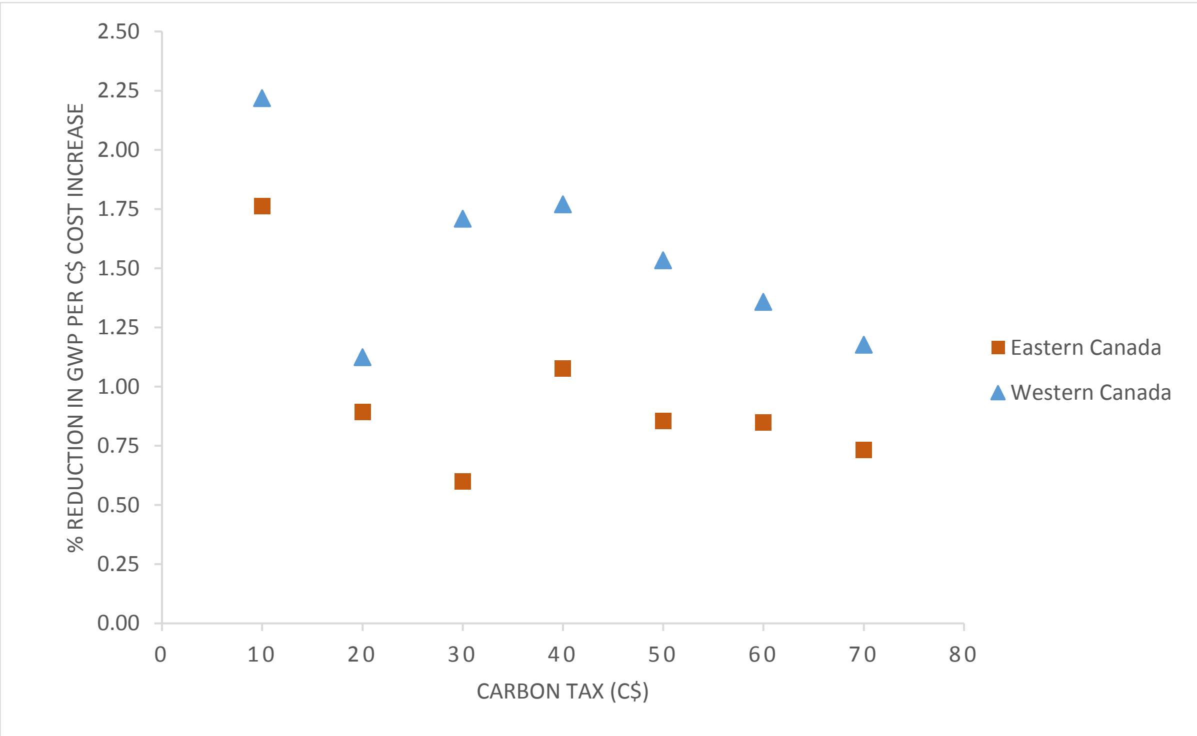


Figure 6.5 The relative reduction in GWP per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of carbon tax compared to a no tax in scenarios modelled for Eastern Canada and Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq). C$ = Canadian Dollars

6.4.2 Nitrogen tax

The ingredient compositions of the diets formulated at different levels of taxation are shown in Figures 6.6a and 6.6b for the Nitrogen tax. The relative levels of feed cost, feed intake, N excreted, P excreted, NRRU, AP, EP and GWP compared to no taxation shown in Figures 6.7a and 6.7b for the Eastern and Western Canada scenarios respectively. In both regions the N tax was unable to produce significant reductions in any of the impact categories caused by the production system through dietary change.

In the East, predicted N excretion decreased as the levels of N tax increased, and was reduced by a maximum of 8% in the highest tax scenario. The tax added to feed costs incrementally as penalty levels increased up to a maximum of 21% at C$$3 per kg N spread. P excretion remained unchanged for all tax levels except at C$3 per kg N spread, when it dropped by 5% compared the no tax scenario. While the N tax worked as a mechanism to reduce N excretion in the scenarios tested, this did not result in any significant reductions in the overall levels of any impact category calculated by the LCA. This was because the changes in the ingredient composition of the diets which caused the reduction in predicted N excretion marginally increased the environmental impacts of the diet per kg. The nutritional density of the least cost diets remained relatively constant at all levels of N tax in the East, with predicted feed intake dropping by no more than 1% compared to the no tax scenario. As N tax increased, the inclusion of ingredients with relatively low levels of environmental impact per kg (see Table 6.2), such as wheat shorts and soybean meal reduced. Levels of corn, wheat and synthetic amino acids in the G/F diets (which were associated with higher impact levels per kg) all increased in order to reduce the crude protein level and amino acid content of the diet and minimise N excretion. The effect of the changes in ingredient composition in increasing levels of AP and EP caused by the diet was such that reductions in these impact categories due to lower N excretion did not translate into reductions in the overall level of AP and EP.

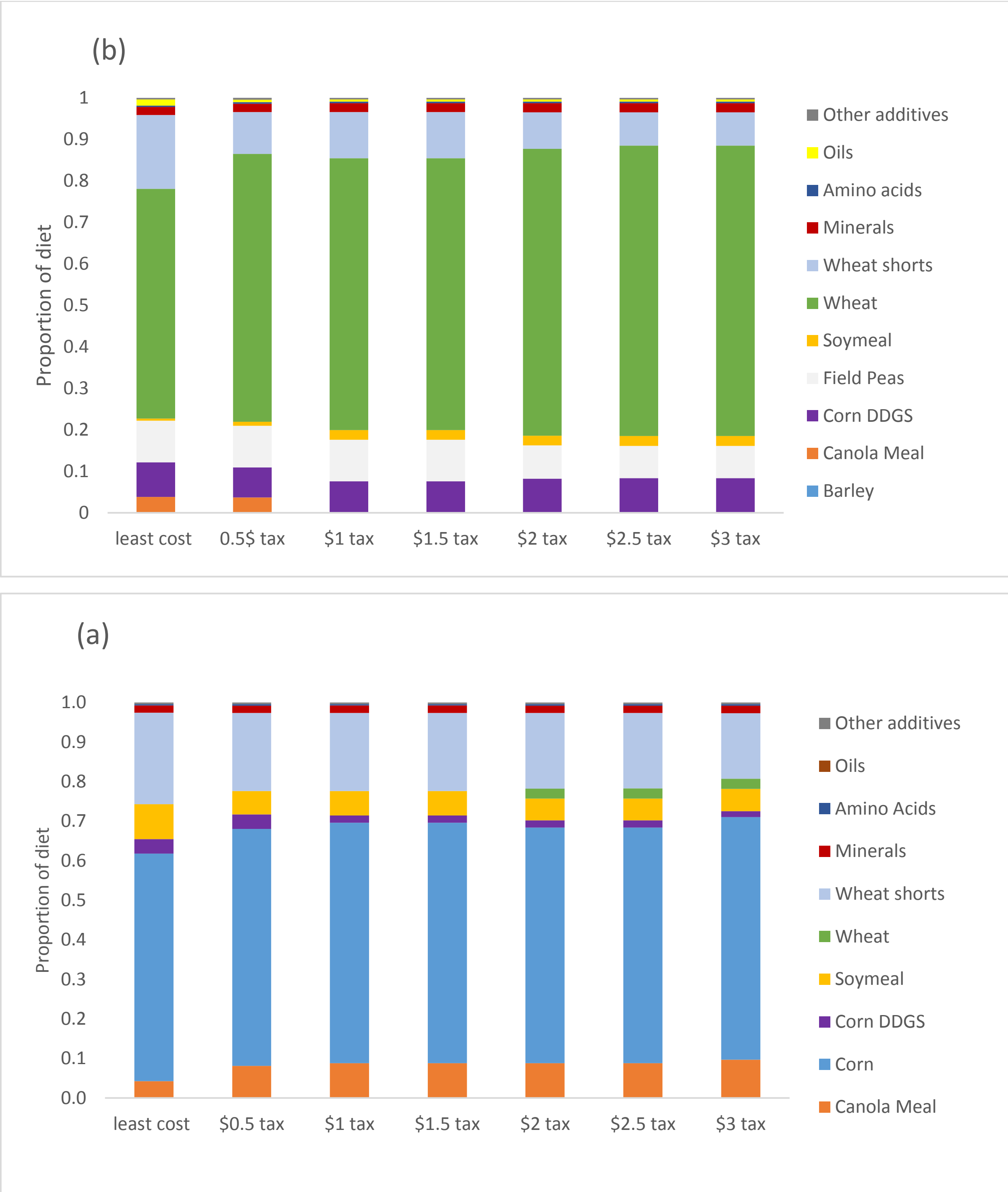


Figure 6.6 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field in manure. C$ = Canadian Dollars.

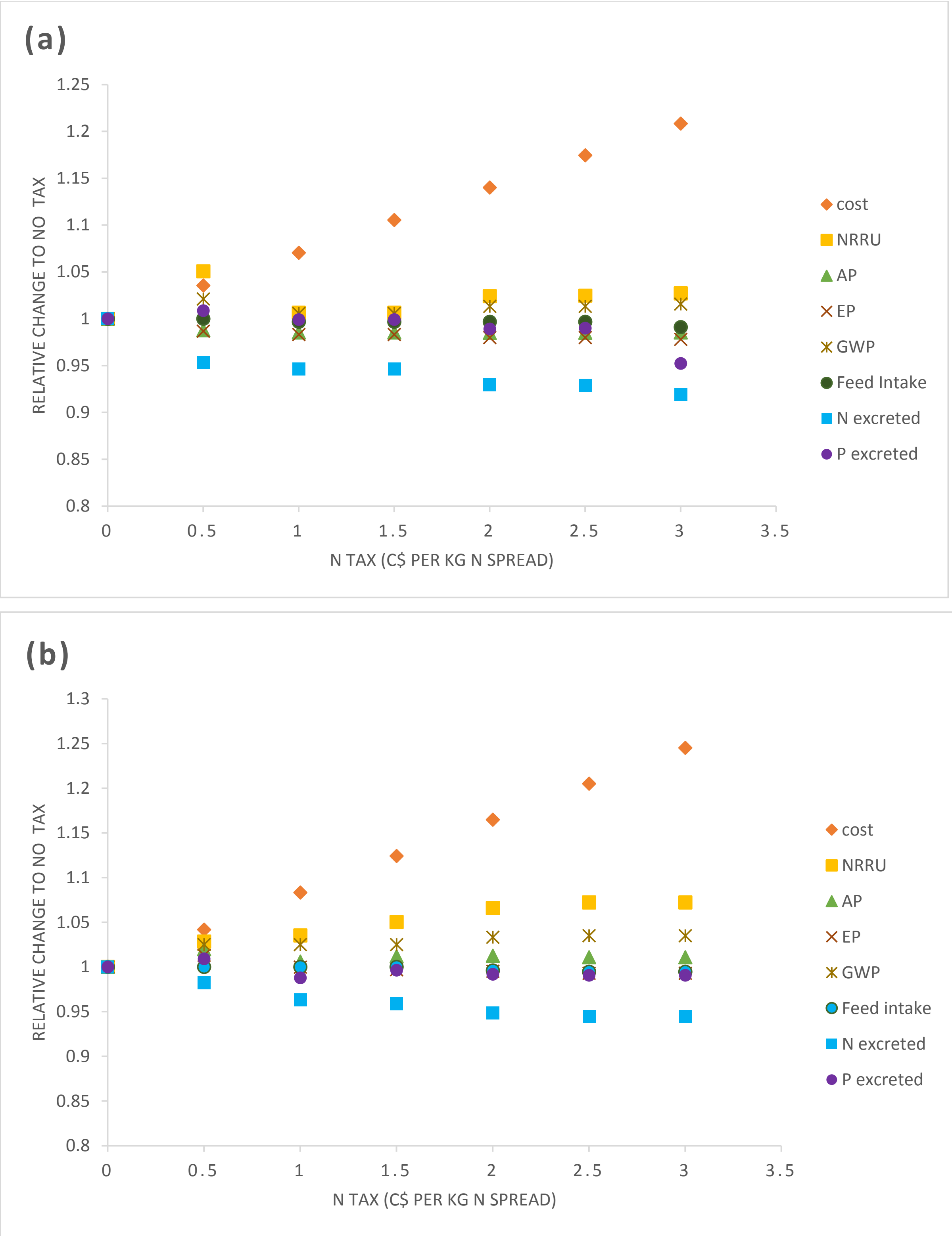


Figure 6.7 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field in manure NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. C$ = Canadian Dollars.

In the West predicted N excretion reduced as N tax increased, with a maximum reduction of 6% compared to the no tax scenario. The cost of feed + tax penalty increased by between 4-25% as tax levels incrementally rose. Predicted P excretion remained similar for all tax levels compared to the no tax scenario. The N tax had little effect on the nutritional density of the least cost diet, with predicted feed intake remaining similar throughout. However, almost all levels the N tax increased GWP and NRRU ($P<0.01$) in the West Canadian scenario, with no significant difference in AP and EP compared to the no tax scenario. As the N tax increased, the inclusion levels of wheat, soybean meal, synthetic amino acids and animal-vegetable oil all increased, with reducing the inclusion of canola meal, field peas and wheat shorts. These changes increased the environmental impact of the diet per kg as fed which negated any reduction in AP and EP as a result of decreased N excretion and actually increased overall levels of NRRU and GWP.

Figure 6.8 shows the relative reduction in N excretion per C$ cost increase for the range of taxes tested in the East and West Canadian scenarios. In both regions, as N tax increased the marginal reduction in N excretion per C$ cost increase was lower, i.e. as N tax increased it became less cost effective at reducing N excretion. At all tax levels greater reductions in N excretion per C$ increase in costs were observed in the scenario for Eastern Canada than that for Western Canada. The relative reductions in N excretion were larger compared to the no tax scenario in the East than the West (see Figures 6.7a & 6.7b), and the N tax was able to make greater reductions in N excretion at a lower relative increase in cost in the East Canadian scenario.

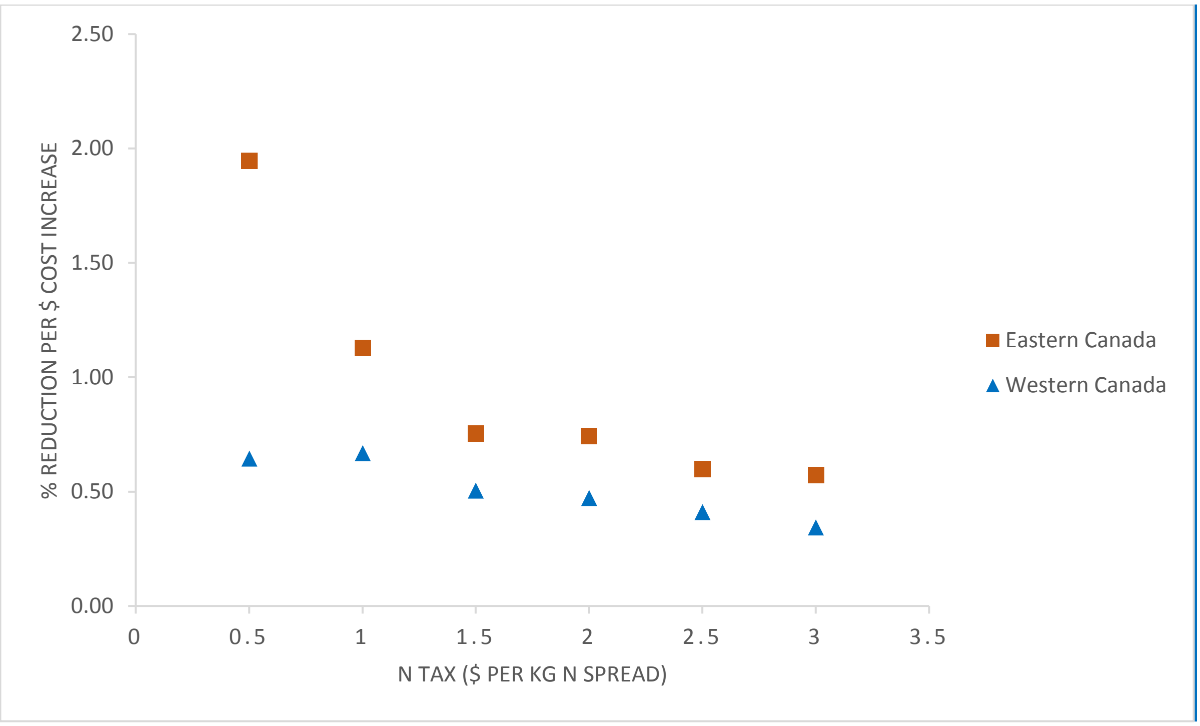


Figure 6.8 The relative reduction in N excretion per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of N tax compared to no tax for scenarios in Eastern Canada and Western Canada. N tax levels are shown in C$ per kg of N in manure spread to field. C$ = Canadian Dollars.

6.4.3 Phosphorus tax

The ingredient compositions of the diets formulated at different levels of P taxation are shown in Figures 6.9a and 6.9b for the carbon tax. The relative levels of feed cost, feed intake, N excreted, P excreted, NRRU, AP, EP and GWP compared to no taxation shown in Figures 6.10a and 6.10b for the scenarios for Eastern and Western Canada respectively.

In the East Canadian scenario the P tax reduced predicted P excretion by a maximum of 22% at tax levels of C$7.5 per kg of P spread and above. The P tax increased the cost of feed + manure spreading by between 4-19% at the increments tested. Predicted N excretion was marginally reduced (by <2%) compared to the no tax scenario at all tax levels. The P tax did not reduce any impact category for all levels of tax tested. Tax levels of C$5 per kg P and above caused increases in NRRU of up to 21% ($P<0.001$) and up to 8% in GWP ($P<0.001$), with no significant difference in AP or EP compared to the no tax scenario. The least cost diets were identical at tax levels above C$7.5 per kg of P spread, as the diet formulation algorithm was unable to alter the diet to reduce costs. The P tax resulted in an increase in the energy density of the least cost formulation and thus a reduction in feed intake of up to 3% in the East above C$5 per kg P. The main alteration to the ingredient composition of the diet was that wheat shorts inclusion (an ingredient with low levels of AP and EP, see Table 6.2) was reduced from 231 g/kg in the no tax scenario to 64 g/kg at P tax levels of C$7.5 and above. The inclusion levels of corn and soybean meal (both higher in AP and EP than wheat shorts per kg, Table 6.2) rose increasing the energy density of the diet and reducing the predicted levels of excreted P. This increased the overall impact levels of the diet per kg as fed, causing the increases in GWP and NRRU and meaning there was no reduction in EP overall in the system despite greatly reduced P excretion.

In the West Canadian scenario, predicted P excretion was slightly reduced by up to 4% within this range of tax levels tested. The cost of feed + manure spreading rose linearly as the P tax increased in the Western scenario at a rate of 4% per increase of C$2.5 per kg P spread. Predicted N excretion was also similar at all tax levels. While the ingredient composition of the least cost diet did change at P tax levels between C$2.5 and C$12.5 per kg P excreted, the tax did not cause significant reductions in any impact category tested in the LCA. Above C$10 per kg of P the tax caused increases in the NRRU resulting from the farming system. At C$15 per kg P excreted the tax did alter the composition of the least cost solution for the G/F diet and reduced predicted P excretion by 11% compared to the no tax scenario. Levels of AP dropped by 6% at this tax level, however, there were increases in NRRU (14%) and cost (24%) compared to the no tax scenario with no significant change in EP or GWP. In the West, the P tax had little effect on the nutritional density of the least cost diet, with predicted feed intake remaining similar for all tax levels. The main alteration to the least cost diet at C$15 per kg P excreted was the inclusion of barley at 140 g/kg, which was not included in the no tax scenario diet. The inclusion of corn DDGS also increased and wheat inclusion was reduced by 150g/kg compared to the no tax scenario. The relative difference between barley and wheat in AP per kg of ingredient (table 6.2) caused the reduction in AP, and increased corn DDGS inclusion increased NRRU.

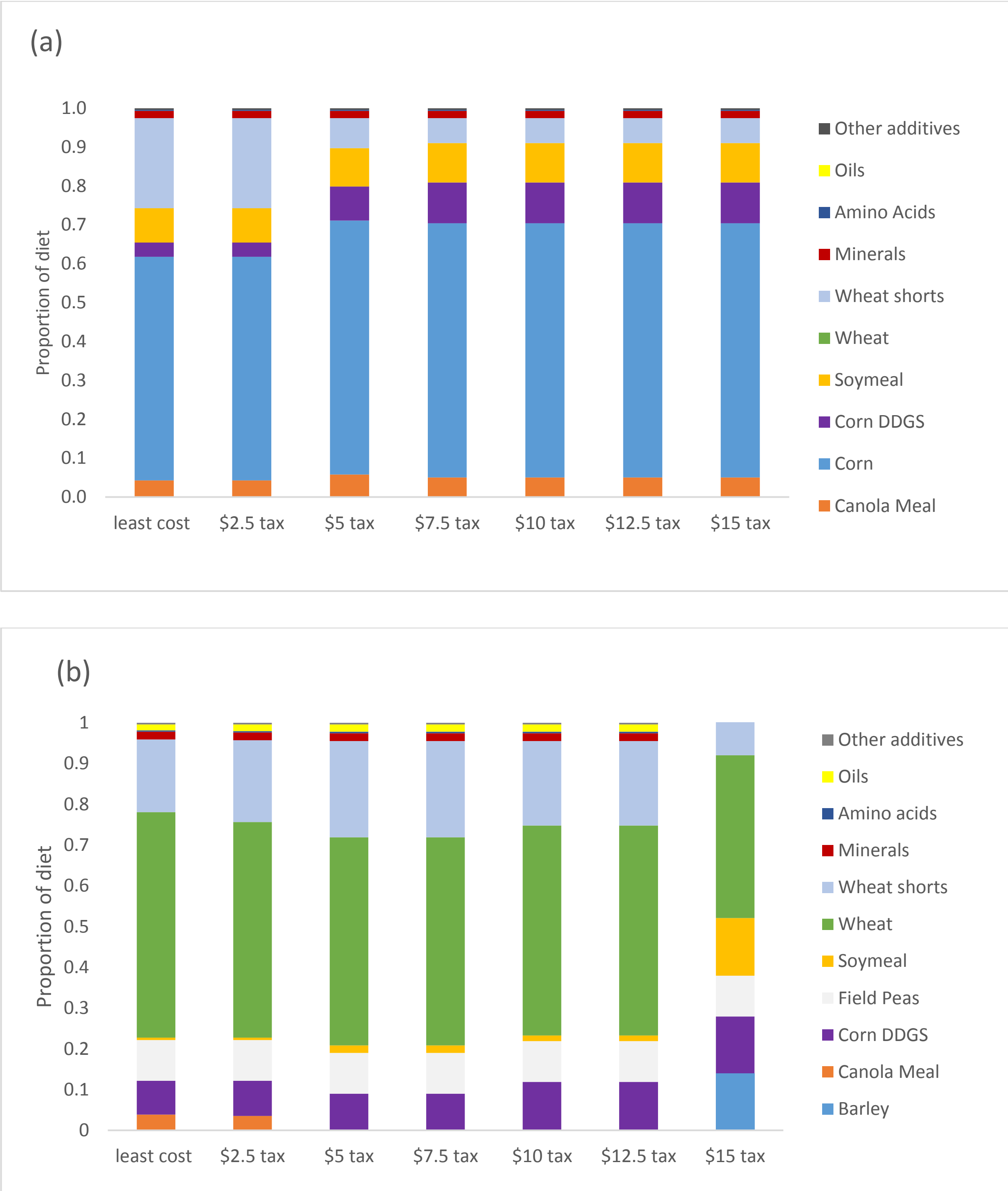


Figure 6.9 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of phosphorus tax in a) Eastern Canada; b) Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in manure.

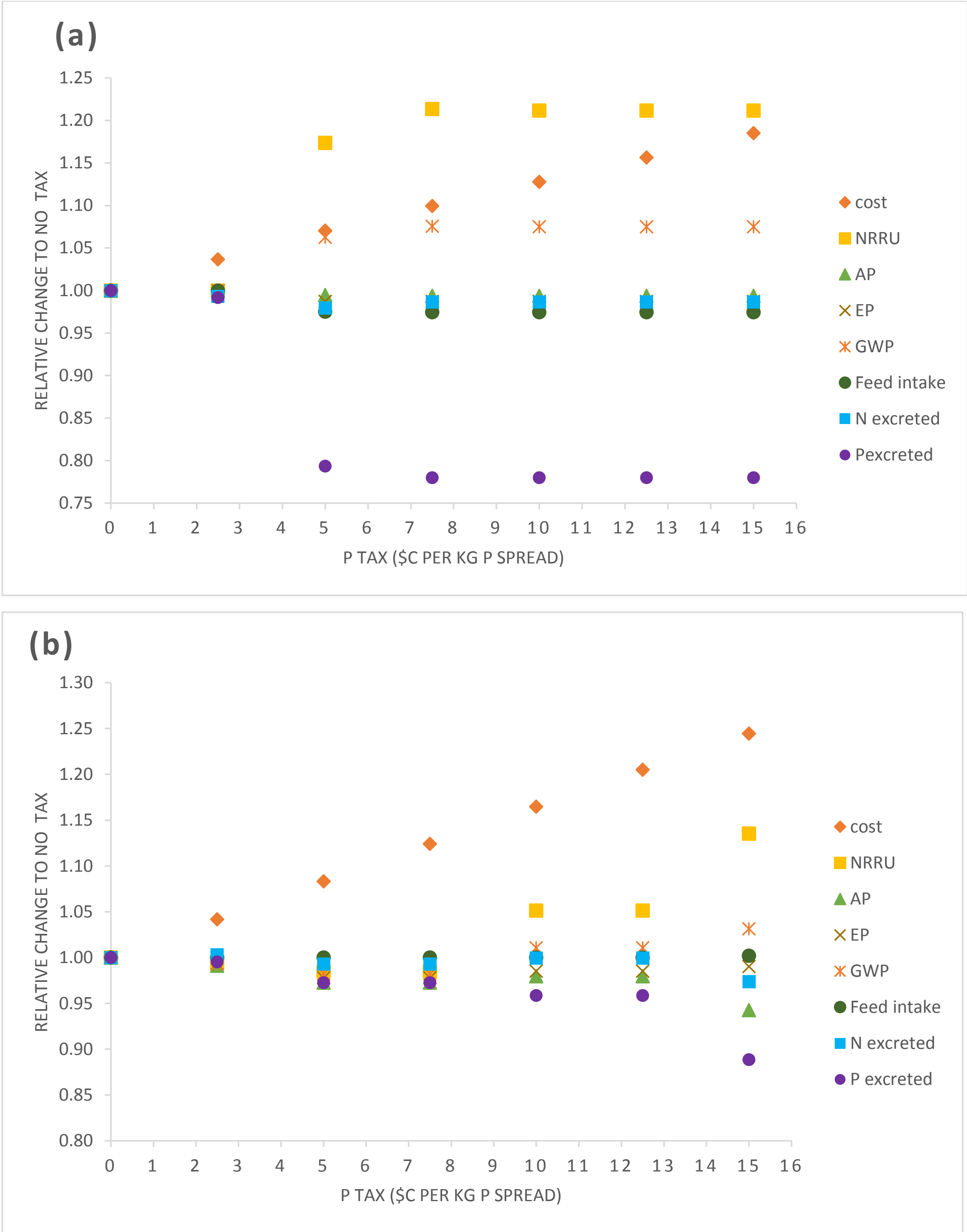


Figure 6.10 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least cost subject to different levels of phosphorus tax in a) Eastern Canada; b) Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in manure NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential.

Figure 6.11 shows the relative reduction in P excretion per C$ cost increase for the range of P taxes tested in the East and West Canadian scenarios. The P tax was able to reduce levels of P excretion in the East Canadian scenario by twice as much as the West Canadian scenario (see Figures 6.10a & 6.10b). As such all tax levels had much larger reductions in P excretion per C$ increase in costs in the scenario for Eastern Canada than that of Western Canada. In the East a C$5 P tax was the point at which largest relative reduction in P excretion per C$ cost increase was achieved, beyond this tax level the relative return on the P tax in terms of reducing P excretion gradually diminished. In the scenario of the Western Canada the highest tax level tested (C$15 per kg P spread to field) produced the greatest relative reduction in P excreted per C$ cost increase. This was because this tax level triggered dietary changes which reduced P excretion by double the amount that any of the lower tax levels were able to achieve.

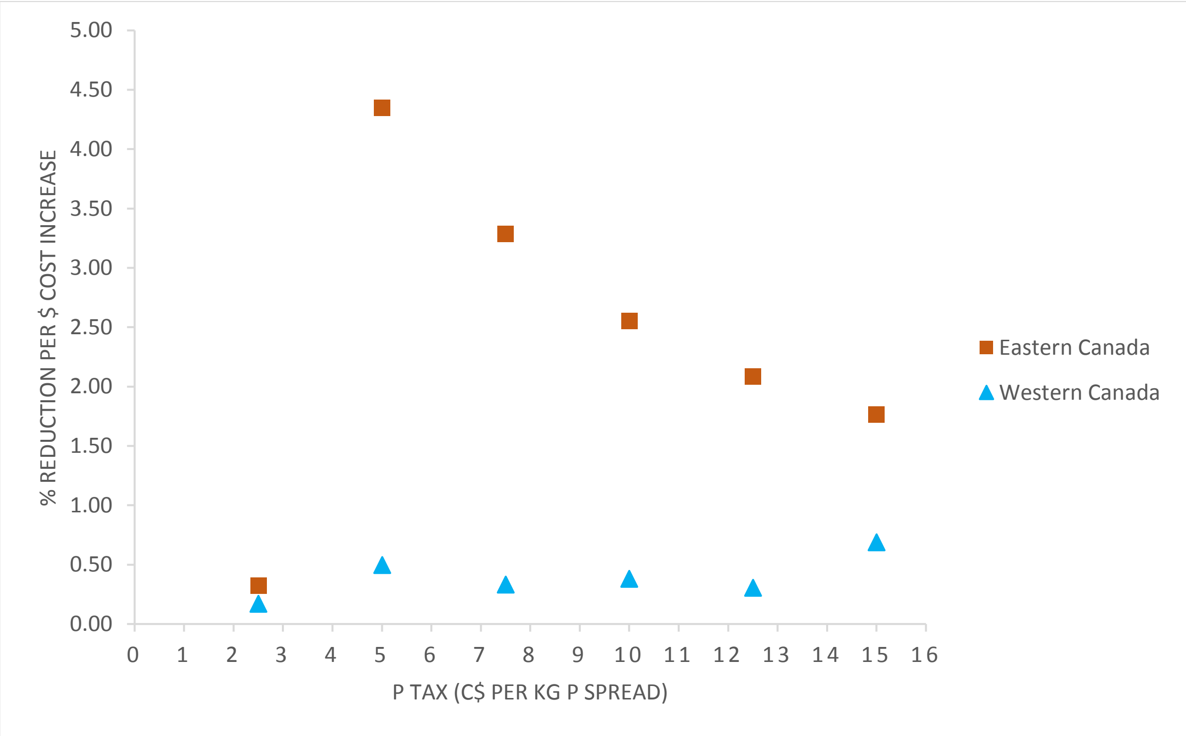


Figure 6.11 The relative reduction in P excretion per C$ increase in feed cost caused by pig diets formulated for least cost and subjected to different levels of P tax compared to no tax for scenarios in Eastern Canada and Western Canada. P tax levels are shown in C$ per kg of P in manure spread to field. C$ = Canadian Dollars.

6.5 General discussion

Greater awareness regarding the environmental impacts of livestock systems, combined with projections of an increased global demand for animal products, has led to increased interest in policy measures to control and minimise these (Steinfeld et al., 2006). The production of feed materials and the storage and disposal of manure are generally the most important considerations regarding the environmental impacts of monogastric livestock production systems (Leinonen et al. 2012; Williams et al. 2006; Basset-Mens & Van Der Werf 2005). The composition of animal diets determines not only the environmental impact of the feed supply chain, but also has effects on nutrient excretion and is thus extremely important in determining the environmental impact of livestock systems. The results above demonstrate that when nutrient excretion is reduced through the introduction of specific taxes, it does not necessarily follow that environmental impacts have been reduced at the system level in livestock farming systems.

Here we investigated the potential effect of 3 different environmental taxes on diet formulation in pig farming systems and the implications for environmental impact. While we use Canadian pig farming as an example system, in any country/region where environmental taxes such as those tested here are implemented, the findings here have broader implications for decision making including diet formulation across the agriculture sector. Expanding the approach to a larger study considering feed decision across livestock systems for different species would provide a more holistic assessment of the effects of such policies. Nevertheless, the more focussed analysis reported here has demonstrated some of the real challenges facing policy makers seeking to reduce the environmental externalities of food production systems.

It was hypothesised that in each case as the environmental taxes increased, they would reduce levels of the targeted emission type i.e. GWP caused by feed production for the carbon tax, N excretion for the N tax and P excretion for the P tax by altering the least cost formulation. As described in the results above this was the case for each scenario tested. This agrees with the findings of previous diet formulation exercises which have integrated levies on P excretion in pig systems (Pomar et al., 2007), as well as a carbon tax on methane emissions in dairy systems (Moraes et al., 2012). However, neither of these studies used an LCA model to assess the implications of dietary change to reduce a specific type of emission for multiple environmental impact categories at the system level. In all of the tax scenarios tested, as levels of tax were increased certain trigger points caused changes in the diet formulated, reducing the target emission and in some cases, environmental impacts at the system level. In some of the scenarios, such as the P tax in Eastern Canada, there were only one or two such trigger points at the lower end of the tax range tested. In these cases it was clear that the optimal level of taxation was at the lower end of the range tested. For the carbon tax and the N tax, the largest reductions in GWP and N excretion per C$ cost increase respectively were achieved at the lowest tax levels tested. The subsequent trigger points for dietary changes as tax levels increased had diminishing returns in terms of cost effectiveness in reducing their target emissions. However in the case of the carbon tax, further reductions in GWP were made as the tax level increased up to C$40 in the East and C$60 in the West. Justification for setting tax rates at these higher levels would be dependent on analysis to value the marginal external cost (MEC) of GHGs. Analysis conducted in this area have produced a wide range of estimates for this depending on different factors, a potential MEC at national level for Canada of around C$42 per tonne CO_2 has been estimated (Anthoff et al., 2009; Waldhoff et al., 2011).

The carbon tax was able to reduce overall levels of GWP for the production system at every tax level tested in both regional scenarios. The carbon tax was assigned to each ingredient based on calculations in the LCA model to determine the GWP per kg in each case. This is a similar framework through which carbon taxes are often implemented within energy markets, by estimating the GWP caused by different energy generation methods (Komanoff and Gordon, 2015). This study optimised animal diets using linear programming to meet nutritional requirements for least cost for different tax levels. The approach has parallels with equivalent exercises for energy markets, whereby the least cost energy generation mix can be determined using linear programming under different tax scenarios (Askar, 2011; Wei et al., 2014). One practical difference between these two scenarios is that there are a much greater number of potential ingredients available to use in animal diets than potential methods of energy generation. This may present an administrative problem for implementation of taxes designed in this manner in the animal feed market. However, LCA databases which quantify the environmental impacts for large numbers feed ingredients at a country or regional level are now being established (Blonk Agri Footprint BV, 2015; Burek et al., 2014), and could possibly be used in such policies.

As shown in the results here, individual policy measures are likely to have spill-over effects, perhaps causing increases in other types of environmental impact. For all of the taxes tested (except the carbon tax in Eastern Canada) at least one tax level tested altered the least cost diet in a way which significantly increased at least one of the environmental impact categories. While they produced reductions in levels of N and P excretion, the N and P taxes were ineffective in significantly reducing any of the environmental impact categories tested in the LCA in almost all scenarios (except at the highest level of P tax in the West). This was slightly surprising given the association between the spreading of these nutrients in pig manure and the impact categories AP and EP. A previous study which formulated G/F diets, for the specific objective of minimising individual environmental impact categories, showed that in the East Canadian scenario modelled here reductions of 5% in AP and 6% in EP were possible, and in the West Canadian scenario reductions of 17% in AP and 10% in EP were possible compared to the least cost diet (Chapter 5). The difference in the outcomes from these two approaches shows that in the model of the pig systems represented here, the manipulation of dietary ingredients to reduce the AP and EP caused by the feed supply chain is more important than reducing N and P excretion. The results emphasise that policy makers should be very clear on their priorities from an environmental impact perspective when implementing environmental taxes on livestock systems. Taxes designed to reduce specific types of pollution, which can most easily be measured, through dietary change are likely to have the unintended consequence of increasing other types of environmental impact caused by the production system. Evaluating potential policies using system level LCA models that account for multiple types of environmental impact, can provide policy makers with a more holistic perspective of the environmental trade-offs involved. This can enable more informed decision making and ensure policies aimed at reducing one type of emission or environmental impact do not undermine other environmental impact priorities.

6.6 **Conclusions**

- Of the three taxes tested, only the carbon tax was consistently effective in producing significant reductions in any of the impact categories tested. As well as this in many cases the tax scenarios increased the levels of some of the environmental impact categories tested. All taxes were effective in reducing the emission which was directly taxed, and Chapter 5 showed it was possible to reduce levels of all impact categories tested here through dietary change. System level environmental impact modelling can give perspective on whether potential environmental taxes are capable of reducing environmental impact in livestock systems.
- Taxes which reduce nutrient excretion through dietary change will not necessarily reduce environmental impacts in livestock systems.
- The largest reductions in the target emission per C$ increase in cost were achieved at the lower end of the tax range tested in most of the tax scenarios. Justification for setting tax

rates at higher levels would be dependent on analysis to value the MEC of target emissions.

- Increased recognition for the importance of reducing GHGs, along with the ongoing expansion of available LCA databases, may cause carbon taxes to be introduced to industries other than energy generation, such as livestock production. This study demonstrated how a potential framework for this could affect diet formulation decision in a narrowly defined scenario. A broader analysis, which simulated how such a tax would affect feed decision across different species in the livestock industry, would increase understanding of the potential implications for environmental impact.

Chapter 7: General Discussion

The environmental impacts of livestock systems have come under greater scrutiny in recent years (Steinfeld et al., 2006). This has meant increased interest in quantifying their environmental burdens and identifying potential solutions to mitigate these using modelling techniques such as Life Cycle Assessment (LCA). Pork is the most consumed form of animal protein globally (Macleod et al., 2013) and is associated with specific environmental concerns. These include resource inputs to the animal feed supply chain and the contribution of nitrogen (N) and phosphorus (P) excreted in pig manure to eutrophication when spread as fertiliser (Basset-Mens and Van Der Werf, 2005; Macleod et al., 2013; Thoma et al., 2013). The thesis had a practical objective; to model the environmental impacts of pig farming systems in Canada using LCA and to quantify the potential for nutritional solutions to reduce the environmental impact of the system. To meet the objectives of this project, methodological issues for LCA modelling of livestock systems regarding co-product allocation, uncertainty analysis in LCA, and the formulation of animal diets for environmental impact objectives also needed to be addressed. The use of the Canadian pork systems was a case in point, as the methodological issues addressed by the project are expected to have application both to pork LCA in general, and to LCAs for other livestock species.

7.1 Modelling the environmental impacts of pig farming systems

The environmental impacts of Canadian pig farming systems were modelled in an LCA with multiple environmental impact categories, using a large industry supplied farm benchmark dataset, for the first time (Chapter 3). The association of the project with an industry partner meant access to very good estimates of key performance characteristics of the pig systems in question, such as mortality rates, litter sizes and gain: feed ratios for different production stages. The mean results for Global Warming Potential (GWP) per kg expected carcass weight (ECW) were slightly lower than the only previous estimate of this for Canadian pig production (Vergé et al., 2009), as well as LCA studies on US pig production (Pelletier et al., 2010; Thoma et al., 2011). However, most of the discrepancies between the findings here and in those studies were explained by a couple of key differences in the assumptions regarding the production system and emission factors (see Chapter 3). Furthermore, the results for overall levels of environmental impact cannot be compared in any meaningful way to the numerous LCA studies of European pig production. The reason for this is that there are too many differences in the background system of an LCA when comparing production systems in different continents. However, the results reported in Chapter 3 broadly agree with most

LCA studies of pig systems studies in showing that; 1) the feed supply chain combined with emissions during housing and manure management were responsible for the majority of environmental impacts caused by pig systems, and 2) The Grower/Finisher (G/F) stage is the production phase where the majority of environmental impact occurs as this requires the most feed. (Basset-Mens and Van Der Werf, 2005; Macleod et al., 2013; Reckmann et al., 2013). This made the potential of altering G/F diets to reduce the environmental impact of pig farming systems a logical area for further investigation.

7.1.1 The potential of nutritional solutions to reduce the environmental impact of the pig farming systems

In Chapters 4 and 5 the potential of dietary change to reduce the environmental impact of pig farming systems was modelled using different approaches. Previous LCA studies that investigated the effect of specific ingredient changes in pig diets on the environmental impacts of pig farming systems have mainly focussed on two areas: 1) the impact of crystalline amino acid supplementation (Garcia-Launay et al., 2014; Mosnier et al., 2011a; Ogino et al., 2013) and 2) the use of alternative protein sources to replace soybean meal in European systems (Eriksson et al., 2005; Meul et al., 2012; van Zanten et al., 2015). In this thesis the environmental impact implications of including specific co-products from the human food supply chain in pig diets, namely bakery meal, meat meal and wheat shorts, were modelled for the first time (Chapter 4). The inclusion of bakery meal and wheat shorts in G/F diets of equivalent nutritional specification reduced environmental impact in the scenarios tested. However, including meat meal and corn dried distillers grains with solubles (DDGS) in equivalent G/F diets increased environmental impact for some categories. For corn DDGS this finding was consistent with previous LCA studies on US (corn based) pig diets (Stone et al., 2012; Thoma et al., 2011). Although it was interesting to note that in Chapter 5, where diets were formulated for regional scenarios in Eastern Canada (corn based diets) and Western Canada (wheat/barley based diets), corn DDGS inclusion increased in diets formulated to minimise AP, EP and the combined environmental impact score the West Canadian scenario. Corn DDGS was associated with relatively low levels of AP and EP per kg ingredient compared to other ingredients in the LCA model (see Table 4.4, Chapter 4). In scenarios where corn is not utilised as the main cereal component of pig diets, the nutritional properties of corn DDGS become more advantageous in providing a solution to meet the pig's nutritional requirements. In Chapter 5 this resulted in increased corn DDGS inclusion in diets to minimise AP and EP, highlighting an advantage of considering the implications of the whole diet for environmental impact rather than making assumptions regarding individual ingredients.

In recent years there has been increased interest in utilising co-products in livestock diets, and increased production of bioethanol has also meant increased production of corn DDGS (Stein and Shurson, 2009). This leads to an interesting question; while analysis here and elsewhere showed including increased corn DDGS in corn based pig diets increased environmental impact, what is the fate of this co-product if it is not utilised in livestock diets? Would they most likely be used as fuel, fertiliser or possibly simply become waste. In a broader sense LCA models can aim to answer such questions by adopting a consequential approach (Ekvall et al., 2016). This issue is discussed further in *7.2.1 Co-product allocation.*

The effect of increased inclusion levels of co-products in G/F diets formulated for economic goals (i.e. least cost per kg live weight gain) on the system level environmental impacts, were also modelled for the first time. Four G/F diets were formulated on a least cost basis at 100%, 97.5%, 95% and 92.5% of the energy density required for maximum feed efficiency (within economic limits). The least energy dense diet contained the highest level of co-products and the most energy dense diet contained the least. The least energy dense diet reduced NRRU by 9% and GWP by 4% when compared to the diet designed for maximum feed efficiency, but increased acidification potential (AP) and eutrophication potential (EP) by <1%. It was shown that increased inclusion of co-products, in G/F diets formulated for economic goals, can produce environmental impact reductions for some environmental impact categories in pig farming systems.

The potential for environmental impact mitigation through optimising G/F diets for environmental impact objectives was quantified in Chapter 5. When optimising diets to minimise the impact for a single category, reductions in the said category were observed in all cases. Relatively large proportional reductions were shown to be possible in overall levels of NRRU and GWP when optimising G/F diets to minimise these impacts individually. However these reductions came at the expense of increases in AP and EP. When diets were formulated to reduce multiple environmental impact categories simultaneously, using a combined environmental impact score, only relatively small reductions were possible in the East (5%) and West (7%) Canadian scenarios respectively. Chapter 5 optimised pig diets for an objective combining multiple impact categories for the first time, and this raised some methodological issues (see *7.2.3 Formulating diets for environmental impact objectives*). In the production system modelled it was not possible to make large percentage reductions in any single environmental impact category through diet optimisation without increasing other types of impact caused by the system.

Within this thesis, the potential of specific ingredients and feeding strategies to reduce the environmental impact of Canadian pig systems was investigated within the framework what was currently available in the animal feed market. The potential of novel feed ingredients such as green biomass, insects, algae and bacteria to reduce the environmental impact of pig production were not modelled, although all of these have been put forward as potential pathways to achieve this (Holman and Malau-Aduli, 2013; Makkar et al., 2014; zu Ermgassen et al., 2016). The barriers to this were simple: firstly lack of available data required to build an inventory for producing these ingredients in an LCA model; this is expected and always difficult to overcome for new production processes. Secondly a lack of peer reviewed studies which demonstrate implications of including such ingredients in pig diets (Holman and Malau-Aduli, 2013; Makkar et al., 2014). However, it may be useful for LCA studies to propose some tentative assumptions regarding new ingredients such as these, in order to scope out the potential of these ingredients to reduce the environmental impacts of pig production as well as other livestock sectors. Current projects such as the EU-funded Feed-a-Gene project are expected to contribute towards this issue, especially as they investigate the potential for inclusion of novel protein sources in pig systems (Feed-a-Gene, 2016).

Another alternative feeding strategy which was not modelled in this project was the potential to increase, (or more accurately re-introduce) the feeding of human food waste (or swill) to pigs. The feeding of swill to pigs has been banned in the EU since 2002 following the UK foot and mouth crisis of 2001 (European Commission, 2002). This practice was also effectively banned in Canada in 2007 to meet the conditions of trade agreements, although the practice was not widespread (Reuters, 2007). However, some have called for the ban to be removed in the EU, suggesting that the potential environmental benefits from reduced land use and food waste going to landfill, outweigh the risks posed from feeding swill assuming regulations are followed (zu Ermgassen et al., 2016). While it is unclear whether any reversal of the ban is likely, South Korea and Japan are given as examples of countries which have safely increased the levels of human food waste fed to pigs in recent years. The implications of re-introducing swill as a feed option into modern pig production systems for their environmental impacts are not well understood (zu Ermgassen et al., 2016). Swill is still fed on a large scale to pigs in China, which produces around 50% of global pork (Giamalva, 2014). Given that this is the case, there is a clear lack of LCA studies on Chinese pig production, with a recent review on the subject (McAuliffe et al., 2016) citing only one such study (Luo et al., 2014). This is a clear research gap for modelling the environmental impact of pig systems globally.

7.2 Methodological issues for LCA modelling of Livestock Systems

7.2.1 Co-product allocation

The allocation of environmental burdens to the outputs of multifunctional processes (co-product allocation) is a key concept across LCA modelling. Recently several new methodologies have been proposed for this aspect of LCA modelling in agricultural systems; allocating environmental burdens from multioutput processes based on "causal" physical relationships between co-products (Ayer et al., 2007; FAO, 2014b, 2014c; International Dairy Federation, 2010; Pelletier and Tyedmers, 2011; Van Der Werf and Nguyen, 2015). In Chapter 2, recent methodological developments in this important area of agricultural LCA modelling were examined, finding several issues with efforts to adopt this approach. It was concluded that allocation based on economic value could be most consistently applied across the feed supply chain, when avoiding allocation was not possible in livestock LCA.

Two components of the system modelled in livestock LCA studies which commonly require co-product allocation, are the feed supply chain and the multiple co-products produced by animals. Methodological considerations regarding the latter are affected by the choice of functional unit and system boundary. For instance, in pig production systems, components of the offal may be utilised to produce materials with economic value such as pâté or faggots which can be considered co-products. However, a LCA study of a pig farming system with a functional unit of 1 kg live weight (LW), and a system boundary from cradle the farm-gate, would not need to consider any allocation of the environmental impacts from the production system between these co-products. As can be seen in chapter 1 (Table 1.1) the most common functional unit used in LCA studies of pig farming systems is 1 kg carcass weight (CW), despite this most pig LCA studies do not allocate any environmental impacts to offal (FAO, 2016b). The primary focus of most pig LCA studies, including the work in this thesis, is the farming system, to reflect this the system boundaries of the LCA in the thesis were cradle to farm-gate. A functional unit of 1 kg CW was used in order to make the outputs of the LCA comparable with the majority of other LCA studies. However, all impacts from the pig farming system were allocated to the carcass and none to offal. To properly account for the extra processing steps required to allocate impacts between the carcass and offal would have required expanding the system boundary beyond the farm-gate to the slaughterhouse and possibly further processing steps. For the work conducted in the timeframe of this thesis, obtaining the extra data required to expand the system boundary and model these further processes was an impractical step, which would have distracted from the focus on the farming system. However, this did have some implications for the results produced here particularly in Chapters 4, 5 and 6 where the environmental impacts of different G/F diets were compared. For diets which included large amounts of bulky ingredients such as corn DDGS or wheat shorts, there was an assumed reduction in carcass yield due to increased gut fill, marginally increasing their environmental impacts per functional unit. Allocation of some impacts to co-products made from offal may have partially off-set this effect. With more time and resources, the LCA model presented in this thesis could have been improved by expanding the system boundary beyond the farm-gate, to include the impact of processing steps at the slaughterhouse and even further downstream in the supply chain.

In this project, the outcomes presented in Chapters 4, 5 and 6 were effectively a result of the methodology for co-product allocation (economic allocation) chosen for the feed supply chain. While it would have been possible to present results in this project for multiple allocation methods, it is questionable what value this have added to this project. Using different allocation rules in LCA studies of agricultural systems has been shown many times to result in different conclusions when comparing scenarios (e.g. Thomassen et al. 2008; Nguyen et al. 2011; Eady et al. 2012; Brankatschk & Finkbeiner 2014). This raises a point of concern, which stems from the general issue of price variability changing the outputs of an LCA model using economic allocation (Ardente and Cellura, 2012). What if environmental policies such as those proposed in Chapter 6 alter the price of ingredients? The concern being that this can create a feedback loop in LCA models, making it impossible to quantify environmental impact reductions or use linear optimisation to reduce the environmental impacts of the system as in Chapter 5. The issue of price variability can be greatly reduced by using multiyear averages of commodity prices, and the added cost to commodities caused by scenarios such as the carbon tax in Chapter 6 can easily be ignored (Guinée et al., 2004). The problem of such taxes indirectly affecting prices, for e.g. increasing the market price of co-products with low carbon footprint using this allocation methodology due to having little economic value, is more difficult to resolve. The specific issue could be eliminated by only using price data from before any such legislation was enacted in the allocation methodology. However, this could become contentious in the long term if very old price data was being used to attribute tax liability.

Adopting a consequential approach in LCA modelling essentially means that the implications of prospective changes to the existing equilibrium in a production system are modelled through a series of “what-if” scenarios (Suh and Yang, 2014). This is seen as fundamentally different from attributional LCA by many researchers, where the aim is to describe the environmentally relevant flows to and from a production system (Curran et al., 2005). Co-product allocation is avoided in consequential LCA using a framework known as system expansion or substitution (Finnveden et al., 2009). System expansion classifies co-products from a system as “determining” or “dependent”, with the environmental impact of the system activities assigned entirely to the “determining product (Weidema and Schmidt, 2010). Functions fulfilled by the “dependent” co-products are credited with replacing the need to produce other materials to perform that function. However, LCA models are rarely wholly consequential or attributional (Suh and Yang, 2014). For example, many attributional livestock LCA studies, including this thesis, have used system expansion to account for nutrients in manure replacing the need for inorganic fertilizers when spread on fields for crop production (Reckmann 2013; Williams et al. 2006; Cherubini et al. 2015).

This poses an obvious question for this thesis and more generally for livestock LCA studies. Is it more appropriate to use a consequential approach when modelling the environmental impact implications of feed ingredient choices? In relation to the results presented in Chapter 4, if using corn DDGS in pig diets is shown to increase some types of environmental impact, nutritionists may ask what the fate of this ingredient is if not used in animal feed? There are however, issues with using such an approach. When utilising co-products such as corn DDGS in animal feed there are a multitude of pathways for such material to be used, if not included in the diets for the particular livestock system modelled. Expanding the model with a “what if” scenario to predict the replacement pathway for a particular ingredient, when this cannot be predicted with any confidence, means the modelling exercise strays further away from using known facts (Heijungs and Guinée, 2007). Many researchers such as Lundie et al. (2007) argue that consequential LCA should not be used in such instances as the uncertainties introduced are greater than those which stem from an attributional approach. It was decided that such a consequential modelling approach was not a viable option for the questions asked in this project.

While the majority of existing animal production system LCA studies adopt an attributional approach to modelling the feed supply chain (De Vries & De Boer 2010), examples do exist modelling pig systems using a consequential approach (Dalgaard et al., 2007; Nguyen et al., 2011), but neither of these modelled feed decision scenarios directly. Van Zanten et al. (2014) used consequential modelling to test the impact of utilising co-products as feed for dairy cattle instead of pig diets (for wheat middlings) or producing bioenergy (for beet tails). Rather than modelling the implications of this for the environmental impacts of the livestock products however, their analysis modelled the effect of this decision on the GWP and land use of the co-products themselves. Such an analysis for the scenarios presented in this thesis would be extremely complex. Further collaboration between LCA modellers and economists however, could allow dynamic market models to be used to answer such questions; at the very least determining which other species is most likely to utilise a feed material if it is not used in a particular production system.

7.2.2 Uncertainty analysis

Developing a new methodology for uncertainty analysis in livestock LCA was one of the key outcomes of this thesis (Chapter 3). Uncertainty analysis has been a neglected area of LCA modelling for agricultural systems due to its complexity and extensive data requirements (Leinonen et al., 2012). However, uncertainty analysis is an important aspect of LCA modelling and is key to the credibility of model outputs, particularly when LCA is being used as a decision support tool (Leinonen et al., 2013). The methodology developed here built on the concept of categorising uncertainty when comparing two or more alternatives as either specific to one of the scenarios (α) or shared between them (β). Previously this approach had been applied by removing shared uncertainty from the scenarios being compared and simulating the distribution of results for these individually (Leinonen et al., 2012). One issue with this approach was that it didn’t include shared uncertainty in comparisons between two production scenarios, thus assuming their calculated environmental impacts were affected evenly by this uncertainty. This may not be the case; for example uncertainty in the predicted yield of a crop included in an animal diet for two production scenarios is shared between these systems, but will cause different levels of uncertainty in the calculation of their impacts if the crop makes up a larger proportion of the diet in one of the scenarios. The uncertainty analysis presented in Chapter 3 utilised the parallel Monte-Carlo simulation function available in SimaPro®. This allowed the LCA to include the effect of shared uncertainty when comparing two scenarios in the LCA, while calculating probabilistically which will result in a greater level of impact for a particular impact category.

However, this approach was not completely without disadvantages. Firstly, Monte-Carlo simulations within SimaPro® are time consuming due to the large databases driving the calculations. SimaPro® is also only compatible with single-thread processing and is slower in performing Monte-Carlo simulations than software such as Matlab® which can use multi-thread processing to perform and repeat calculations at much greater speed (Hoorn, 2009). It was also only possible to compare 2 scenarios at a time using this method in SimaPro®, This significantly increased the time it took to perform the analysis in this thesis, meaning that in Chapters 4, 5 and 6 alternative diet scenarios were only compared statistically to the control diet in each case rather than to all other scenarios modelled due to time restrictions. Again utilising a generic modelling platform designed for Monte-Carlo simulations such as Matlab® would have removed this issue (See *LCA software and Modelling limitations* for further discussion on the choice of software).

An alternative to using Monte-Carlo simulations for uncertainty analysis in LCA is to use error propagation (Groen et al., 2014a; Leinonen et al., 2016). This approach avoids the computational issues associated with Monte-Carlo simulations, but requires the relationships within the LCA model to be aggregated into simplified emission coefficients to reduce the number of calculations as these must be done manually. One disadvantage of this is that normal distributions must be assumed for parameter uncertainty ranges, whereas Monte-Carlo simulations can deal with other types of distribution for e.g. lognormal or uniform (Leinonen et al., 2016). The uncertainty analysis methodology developed in Chapter 3 is particularly adept for comparing scenarios which contain high levels of shared uncertainty, enabling useful decision support in scenarios such as those presented in Chapters 4, 5 and 6. The ability to apply the methodology within one of the most popular LCA software packages should increase its likelihood of application in future livestock LCA models.

7.2.3 Diet formulation for environmental impact objectives

Developing a methodology for the optimisation of pig diets for environmental impact objectives was another important contribution of this thesis (Chapter 5). The algorithm developed formulated diets for an objective which considered multiple environmental impact categories, accounted for the effect of nutrient excretion and allowed flexibility in the nutritional specification of the diets for the first time in livestock systems.

The process highlighted some methodological challenges for introducing environmental impact objectives in a diet formulation algorithm, some of which were not easily resolved.

When formulating diets for environmental impact objectives in livestock systems, adopting a single metric for the objective is necessary in order to optimise diets using linear programming. If multiple environmental impact categories are to be accounted for directly in the objective, a combined environmental impact score must be defined. This is a contentious area within LCA modelling, combining environmental impact characterisation is considered unscientific by some researchers and there has been a reluctance to engage with the issue by groups which work towards method standardisation (Finnveden et al., 2009). This is understandable, as ultimately combining and weighting environmental impacts is a value judgement and values cannot be harmonised or evaluated scientifically. The relatively simplistic methodology used for the combined environmental impact score in Chapter 5, which weighted the four impact categories included equally, demonstrated a scenario where it was considered important to avoid large increases for any impact category. It was beyond the scope and resources of this thesis to advance the discussion on how best to weigh environmental impacts. However, combined environmental impact scores have useful applications in multi-criteria decision making (MCDM) problems such as that presented in Chapter 5 (Finnveden et al., 2009). Further efforts by governments and NGO’s to define environmental impact priorities at national and regional levels would aid researchers in agreeing such methodologies, and thus to develop tools to adapt livestock production practices for reduced environmental impacts.

7.3 Environmental impact characterisation

How the environmental impacts of a system are characterised in an LCA is always a subjective choice; which general types of impact e.g. water use, GWP or Eco-toxicity need to be considered? Which specific methodology should be used in each case? The more environmental impact categories included in an LCA model, the larger the data input requirements for the model and the time required to develop it. The LCA model used in this thesis focused on modelling a few key impact indicators which were identified as of importance for pig farming systems. GWP, AP, EP and non-renewable resource use (NRRU) were modelled in all Chapters of this thesis; GWP, AP and EP are consistently recognised as important impacts of pig production and are most commonly used in LCA studies of pig systems (McAuliffe et al., 2016). Draft stage FAO livestock environmental assessment and performance partnership (LEAP) guidelines for modelling environmental performance of pig supply chains includes the impact categories GWP, EP and non-renewable energy use (NRE) (FAO, 2016b). All of these were included in Chapters 3 & 4; NRE and NRRU are heavily correlated impact categories and it was not felt necessary to include both in the latter Chapters of the thesis.

However, there were some important environmental impact considerations for livestock systems which were not included; freshwater use is the other impact category included LEAP guidelines for pig systems (FAO, 2016b), but was not included in the studies presented in this thesis due to limited time and resources. Agriculture is thought to account for around 92% of global human freshwater use, with livestock systems using 27% (Hoekstra and Mekonnen, 2011). It is estimated that pig production systems account for around 5% of global freshwater consumption (Mekonnen and Hoekstra, 2012). Crop irrigation accounts for a large proportion of global water use and it is generally accepted that feed choice is the most important factor in determining the water footprint of livestock products (Gerbens-Leenes et al., 2013). Life Cycle Inventory databases for agriculture such as Agri-footprint are starting to integrate country specific water use data for crop systems, to aid researchers in calculating water footprints for livestock products (Blonk Agri Footprint BV, 2015). Agri-footprint now also utilises the regionalised approach to water footprinting in LCA set out by (Pfister et al., 2009). While this methodology does not rank water stress as a key concern for production systems in Canada, diet formulation algorithms such as that presented in Chapter 5 which integrate water footprint data, could be utilised to reduce the water footprint of livestock systems in regions of greater freshwater scarcity.

7.4 LCA software and modelling limitations

LCA modelling often requires large datasets, modelling hundreds or thousands of small processes which make up the system being modelled. Data requirements can be particularly large when analysing agricultural products due there complexity and high levels of interconnectivity between different aspects of agricultural systems (Audsley et al., 1997; Leinonen et al., 2016; Lundie et al., 2007). Commonly, the large data requirements of LCA modelling are dealt with by collecting primary data for the foreground system which is the focus of the LCA while using existing average data for the background system (Hospido et al., 2010). Specialised LCA modelling software packages such as SimaPro® or GaBi® incorporate large databases of inventory data, such as Eco-Invent which enable users to easily build background systems for their LCA models (EeB Project, 2012). This is one of the main advantages of using such software and makes them ideal for a project such as this thesis where time and resources were limited. However, this convenience also comes with drawbacks from a modelling perspective; a key disadvantage often being fragmentation between the LCA model and some sub-models which describe important aspects of system behaviour. The main reason for this is the limited number of mathematical functions and restrictive interfaces which are provided by these bespoke LCA software packages. For example, for this project it was not possible to use linear programming to formulate diets within SimaPro, neither was it possible to integrate an animal growth model with the LCA, as the necessary functions were not available in the software. When performing system level modelling, ideally all aspects of a model (i.e. its sub-models) should be as integrated as possible to ensure that any effects from changes to one aspect of the system are properly reflected across the system.

Some LCA models of livestock systems have been produced in generic modelling languages such as Python, GAMS or VBA (Williams et al. 2006; Leinonen et al. 2012; Macleod et al. 2013; Burek et al. 2014; Garcia-launay et al. 2015). These models have been able to utilise the advantages of these platforms to integrate elements such as animal or crop growth models directly in the LCA. In some cases they have used this flexibility to create publically available tools which enable external users to calculate the environmental impacts of livestock farming scenarios (National Pork Board and University of Arkansas, 2013; Vellinga et al., 2013). Developing such tools is a time consuming and potentially expensive process that was not possible for a project of this scale. However, where possibly it is desirable that such LCA models are made available in this way to encourage understanding of and engagement with environmental impact issues in the livestock industry.

Another issue when using specialist LCA software is the limited methodological choices this provides for sensitivity analysis; a fundamental aspect of developing any quantitative model (Saltelli et al., 2008). In this project a local sensitivity analysis was conducted using the one factor at a time approach during the LCA model development (Chapter 3). For large complex modelling exercises such as LCA, a global sensitivity analysis which can account for parameter correlations to identify the true sources of variance in model simulations is probably more appropriate (Groen et al., 2014b; Wei et al., 2015). Global sensitivity analysis functions not currently available in most popular LCA software tools, but both GaBi® and SimaPro® have the capability to perform Monte Carlo simulations . This means the software tools are capable of performing matrix based LCA calculations which would be necessary incorporate global sensitivity analysis (Wei et al., 2015). Effort by software providers to rectify this would aid improved LCA model development.

7.5 Utilising animal growth models in diet formulation

One aspect of this project which would have benefited from more time and resources was the algorithm used to formulate pig diets in Chapters 4, 5 and 6. Diets were formulated in this thesis using linear programming to meet a deterministic set of nutritional requirements for pig growth across each feeding phase. Integration of a more mechanistic animal growth model, which considered animal response to diets on a daily basis, with the diet formulation tool would have allowed different questions to be asked when optimising of the dietary regime. These might include the effect of the number of feeding phases, or in a stochastic model, adapting diets to meet the daily nutritional requirements of only a certain percentage of pigs. Commercial tools which integrate mechanistic animal growth models are also been able consider the potential implications of other environmental factors such as stocking density, sorting pigs in different pens according to weight, and health status for animal performance (Ferguson, 2014). If integrated with an LCA model, such a tool would be able to investigate the implications for environmental impact for different approaches in these aspects of the production system A good example of the potential advantages of such an approach was presented by Garcia-launay et al. (2015). Using an integrated diet formulation – animal growth model based on the principles of Brossard et al. (2009), they formulated pig diets for an objective which considered both economic performance and the contribution of the system to climate change. The integration of an animal growth model, and the use of a non-linear optimisation algorithm for diet formulation, allowed that study to formulate diets based on predicted performance at the herd level. This allowed the implications of altering the number of feeding phases used for a grower/finisher (G/F) feeding regime to be modelled. The growth modelled the response of herd performance to different diet formulations, adjusting the length of each feeding phase accordingly and iteratively reaching the optimal solution for a particular objective. It was not possible to develop a mechanistic growth model and integrated diet optimisation tool for this project within the time available while meeting the other requirements to build an LCA model. Further efforts to integrate animal growth and LCA models, which can account for multiple environmental impacts, could enable different approaches to reducing the environmental impact of livestock systems through dietary change to be identified than those presented in this thesis.

One potential area for further development in formulating livestock diets is accounting for how variability in the nutritional composition of ingredients affects animal performance. Presently commercial diet formulation tools tend only to model stochasticity in animal performance traits to simulate herd variability, deterministically describing the nutritional characteristics of feed ingredients (St-pierre and Weiss, 2012). The effect of variability in the nutritional characteristics of the feed is limited in diet formulation tools by restricting inclusion levels of highly variable ingredients, as was done in this thesis. Concerns regarding ingredient variability are often cited by nutritionists as a barrier to increasing the inclusion of alternative ingredients in animal diets, such as the co-products investigated in Chapter 4 (Bogges et al., 2008; Zijlstra and Beltranena, 2013). Recent research has suggested that variability in the characteristics of feed ingredients may more important in influencing variability in pig performance than pig characteristics, when feeding diets with high levels of co-products (Symeou et al., 2016). This has implications for efforts to increase the inclusion of alternative ingredients in animal diets to reduce environmental impact. Further effort to account for the effects of ingredient variability on animal performance in animal growth modelling will enable the appropriate inclusion levels of alternative ingredients in livestock diets to be determined more systematically. These levels may be different when formulating diets for environmental impact objectives in comparison to commercial ones.

Models which can describe the genetic characteristics of animal populations could also be used to model the potential implications of genetic change and selective breeding for the environmental impacts of livestock systems. Some LCA studies have performed retrospective analysis, to identify how the environmental impact of livestock production systems have changed over the years (Boyd et al., 2012; Pelletier et al., 2014). These studies cite genetic change as a key factor in driving improvements in feed efficiency and reductions in the environmental impacts per kg of product. LCA models which can predict animal response based on genetic characteristics to potential feeding strategies, or changes to production practices for enhanced welfare, may play an important role in identifying socially acceptable animal production systems which minimise environmental impact.

7.6 Sustainability modelling – wider issues for pig farming systems

This thesis focussed on how to model and improve the sustainability of pig farming systems in terms of their environmental impacts. A holistic approach to measuring sustainability requires the consideration of the environmental, economic and social implications of an activity or industry (Brundtland, 1987; Morelli, 2011). Modelling the latter two aspects of the sustainability triangle in pig farming systems extensively was not the primary aim of this thesis, although in Chapters 5 & 6 the predicted cost of the diets formulated was presented for in the analysis. LCA studies such as this one can be integrated into wide ranging assessments of sustainability in the livestock sector which consider all three pillars of sustainability (Bonneau et al., 2014; Dourmad et al., 2014). Use of the ecosystems services modelling framework is becoming more widespread in efforts to holistically model the sustainability of livestock systems (Chatterton et al., 2015). The framework classifies services provided by an ecosystem to humans into four categories: provisioning, regulating, cultural and supporting. Recent analysis of the UK livestock sector under this framework suggested the main benefits of the sector came from provisioning (i.e. producing products such as milk and meat), as well as cultural benefits. These benefits were considered to outweigh the problem of emissions eroding the regulation of other ecosystems relied on by humans, but only if employment was classified as a provision provided by the sector (Chatterton et al., 2015). Analysis frameworks such as this are very useful in providing some structure to the complex task of trying to quantify sustainability holistically.

In livestock production there are important social considerations regarding animal health, welfare and safety in the human food supply chain (Bonneau et al., 2014), as well as the usual economic and social considerations applicable other industries. Another social consideration which is often discussed regarding livestock production in popular debate is the concept of its net contribution to human edible food (Council for Agricultural Science and Technology, 2013). Concerns regarding future food security mean that use of human edible food is an important ethical concern for the sustainability of the livestock industry (Eisler et al., 2014; Pimentel and Pimentel, 2003; Steinfeld et al., 2006; Wilkinson, 2011). Several studies have suggested that when thinking of feed efficiency in terms of human edible food, for e.g. MJ human edible energy contained in feed / MJ human edible energy output, modern non-ruminant production systems compare unfavourably to ruminant systems based on grazing (Dijkstra et al., 2013). This is not surprising, given that monogastric diets are normally heavily based on cereals and vegetable proteins which are potential human food sources (Poulsen et al., 2013). However, such analysis does not fit with the common narrative on sustainability concerns for the livestock sector that non-ruminant production systems are of secondary concern to ruminant systems, as they have better feed efficiencies and lower carbon footprints (de Vries and de Boer, 2010; Macleod et al., 2013).

Wider awareness of climate change issues, by the policy makers and the general population has driven a volume of important research on how to reduce greenhouse gas emissions from livestock systems, a particular issue for ruminant production (de Vries and de Boer, 2010; Eshel et al., 2014; Steinfeld et al., 2006). Similarly, it is possible that the pressure to reduce human edible food used in non-ruminant productions systems may become greater as food security issues become tangible with a wider audience. Statistics regarding the amount of food required to produce 1 kg of meat or protein in meat (which animal scientists would recognise as traditional measures of feed efficiency) are often used by NGOs and pressure groups to discourage the consumption of animal products (e.g. Andersen & Kuhn 2014). Methodologies to define human edible food and how much of it is contained in common feedstuffs are not well developed, but would be essential to provide quantitative analysis in this area. The main issue in this being the difference between what humans could eat and what they will choose to eat which is subject to many social factors. While it would not strictly be an environmental impact category, human edible food input could easily be integrated into conventional LCA frameworks for livestock systems. The result could simply be presented as a resource input, like land use or water use per functional unit produced. Livestock diets could be formulated to minimise the competition between human and animal feed supply chains in the same way as the environmental impact objectives shown in Chapter 5. However, agreeing an accepted methodology to classify exactly how much human edible food is contained in animal feed stuffs is currently a barrier to this.

7.7 The potential effect of environmental taxes on environmental impact from livestock systems

A slightly different question was posed in Chapter 6, which examined whether environmental taxes could be used to drive dietary change to reduce the environmental impact of pig farming systems. Of the taxes tested, only the carbon tax was consistently effective in producing significant reductions in any of the impact categories tested. This highlights a potential issue for policy makers as all taxes were effective in reducing the emission which was directly taxed. While they produced reductions in levels of N and P excretion, the N and P taxes were ineffective in significantly reducing any of the environmental impact categories tested in the LCA in almost all scenarios. The contrast between this and the diets which minimised AP and EP in Chapter 5 showed that AP and EP were reduced more effectively by altering the ingredient composition of the diets than reducing N and P excretion. Modelling the implications of tax scenarios for the environmental impacts of pig farming systems in Chapter 6 demonstrated the potential for the wider application of such LCA models in the area of environmental policy. A renewed focus on the contribution of livestock production to GWP following the Paris climate change summit in 2015 has recently led a Danish think tank to recommend implementing a carbon tax on livestock products in order to alter eating habits

(Withnall, 2016). It will be important that the implications of such taxes for decision making in the food supply chain are modelled using LCA to show they are having the desired effect.

7.8 Scope for future research

During the course of this project, several potential future research objectives for modelling sustainability in pig and more generally livestock systems were identified:

- Value could be added to future LCA models of livestock systems from further integration with mechanistic animal models, which define key characteristics regarding growth and the response of animals to changes in their feed and environment. These may include the implications of precision feeding or sorting practices for the environmental impacts of the system. In some cases LCA models have now started to include animal growth models which predict feed intake for a limited range of circumstances (Garcia-launay et al., 2015; National Pork Board and University of Arkansas, 2013). Defining animal characteristics could allow LCA models to answer important questions regarding the implications of genetic change in livestock animals for environmental impact. This may be in relation to feeding strategies the industry will need to adopt in the future, but also the potential implications of other major changes which may be necessary for the sustainability of livestock production systems; for e.g. a ban on using antibiotics.

- The amount of human edible food used in animal feed is an important ethical concern for the sustainability of the livestock industry, particularly for non-ruminants (Eisler et al., 2014; Pimentel and Pimentel, 2003; Steinfeld et al., 2006; Wilkinson, 2011). An agreed methodology for what constitutes human edible food, and analysis of how much of it is contained in common feedstuffs are currently lacking. Efforts to develop one would be beneficial to enable researchers to provide quantitative analysis of exactly how much competition there is between the human and livestock food supply chains, which remains unclear (Council for Agricultural Science and Technology, 2013). Such a methodology would also enable researchers to quantify how much this could be reduced through alternative feeding strategies.

- While animal growth models are now able to account for variability in animal characteristics (Pomar et al., 2003), accounting for variability in the properties of ingredients has proved more difficult. Improving how animal models deal with this will be enable the implications of using alternative ingredients, which cannot be used for human feed, to be modelled with more confidence. Greater understanding regarding the implications of variability in ingredient characteristics for animal performance, would enhance the identification of sustainable feeding strategies in livestock systems. This could also aid in the early stage modelling of the potential to include novel feed ingredients such as bacteria or insect protein in pig diets before their nutritional properties are well understood. Thus scoping the potential benefits of these ingredients in terms of reducing environmental impact at an earlier stage.

- When reviewing existing LCA studies on pig farming systems it was interesting to note the lack of almost any study which modelled the environmental impacts of Chinese pork production. A recent review on the subject (McAuliffe et al., 2016) confirmed this, citing only one such study which focused on manure management techniques (Luo et al., 2014). This appears to be an obvious knowledge gap in relation to the environmental impact of pig farming globally; China produces and consumes around 50% of global pork (Giamalva, 2014). Breeding and feeding practices from industrial production systems in Europe and North America are becoming more commonplace within the Chinese pig sector (Rabobank International, 2012; Schneider and Sharma, 2014). One of the many effects of this will be to significantly reduce the amount of human food waste fed to pigs in China, increasing the amount of crops fed globally to pigs. The social and environmental implications of industrialising the Chinese pig sector for the sustainability of the global food system warrant further investigation.

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Appendix A1: Composition of typical diets for pig systems in Eastern and Western Canada (Chapter 3)

Table A1.1 Eastern Grower/Finisher diets – Ingredient and Nutritional composition (as fed)

Ingredient		Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Finisher 2 (g/kg)
Corn		325.0	406.0	432.0	441.0
Soybean meal high protein		130.0	67.0	16.0	23.0
Canola meal		100.0	120.0	129.0	147.0
Corn DDGS		100.0	125.0	125.0	125.0
Bakery product		35.0	40.0	57.0	150.0
Wheat		150.0	150.0	50.0	26.0
Wheat shorts		84.0	41.0	150.0	40.0
Animal-Vegetable fat (mix)		41.0	22.0	17.0	18.0
Limestone		14.4	13.5	11.0	11.5
NaCl		4.6	3.9	3.8	3.8
Additives		16.0	11.6	9.2	14.7
Nutrient Name	**Units**	**Starter (g/kg)**	**Grower (g/kg)**	**Finisher (g/kg)**	**Finisher 2 (g/kg)**
Dry matter	%	87.8	87.4	87.2	87.2
Protein	%	19.1	17.4	15.8	16.4
Total P	%	0.54	0.50	0.46	0.46
Total K	%	0.63	0.58	0.55	0.56
GE	MJ/kg	17.6	17.4	17.4	17.5
DE	MJ/kg	15.4	15.1	15.1	15.2
ME	MJ/kg	13.8	13.6	13.6	13.7

Table A1.2 Eastern Breeding and Nursery diets – Ingredient and nutritional composition (as fed)

Ingredient		Nursery 1 – 5kg/pig (g/kg)	Nursery 2 (g/kg)	Gestation (g/kg)	Lactation (g/kg)
Corn		443.2	436.0	467.0	477.5
Soybean meal high protein		242.0	203.0	38.0	179.0
Canola meal		25.0	82.4	0.0	0.0
Corn DDGS		0.0	0.0	100.0	50.0
Canola		0.0	0.0	70.0	60.0
Wheat		100.0	185.0	0.0	100.0
Wheat shorts		0.0	0.0	300.0	71.0
Animal-vegetable fat (mix)		44.0	40.0	0.0	23.0
Limestone		5.7	15.5	15.0	17.0
NaCl		4.6	3.9	4.5	4.8
Whey		61.0	0	0.0	0.0
Meat meal		45.0	21.5	0.0	0.0
Additives		29.5	12.8	5.5	17.7
Nutrients	**Units**	**Nursery 1**	**Nursery 2**	**Gestation**	**Lactation**
Dry matter	%	88.1	87.7	87.4	87.2
Protein	%	20.4	19.4	14.2	18.6
Total P	%	0.60	0.67	0.52	0.61
Total K	%	0.85	0.72	0.64	0.74
Gross Energy	MJ/kg	18.5	17.9	16.7	17.9
Digestible Energy	MJ/kg	16.1	15.6	14.5	15.6
Metabolisable Energy	MJ/kg	14.5	14.0	13.1	14.0

Table A1.3 Western Grower/Finisher diets – Ingredient and nutritional composition (as fed)

Ingredient		Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Finisher 2 (g/kg)
Wheat		509.3	459.2	445.4	423.1
Peas		150.0	61.0	0.0	102.0
Canola meal		100.0	39.0	13.0	33.0
Wheat shorts		91.0	142.0	155.0	177.0
Corn DDGS		75.0	50.0	27.0	42.0
Barley		22.0	215.0	328.0	191.0
Limestone		15.0	16.0	13.0	13.0
Soybean meal high protein		12.0	0.0	0.0	0.0
Pork meal		12.0	0.0	0.0	0.0
Potash salt		4.1	4.4	4.6	4.6
Lysine		3.0	3.0	3.0	3.0
Methionine		2.4	2.2	1.6	2.3
Animal Vegetable fat (mix)		3.0	3.0	3.0	3.0
Additives		1.3	3.3	4.6	4.3
NaCl		0.0	2.0	1.9	1.8
Nutrient Name	**Units**	**Starter (g/kg)**	**Grower (g/kg)**	**Finisher (g/kg)**	**Finisher 2 (g/kg)**
Dry matter	%	89.3	89.0	88.9	88.9
Protein	%	20.0	16.4	14.8	16.6
Total P	%	0.54	0.51	0.49	0.51
Total K	%	0.66	0.58	0.55	0.56
Gross Energy	MJ/kg	16.1	15.5	15.3	15.4
Digestible Energy	MJ/kg	14.0	13.5	13.3	13.4
Metabolisable Energy	MJ/kg	13.4	12.9	12.8	12.9

Table A1.4 Eastern Breeding and Nursery diets – Ingredient and nutritional composition (as fed)

Ingredient		Nursery 1 – 5kg / pig (g/kg)	Nursery 2 (g/kg)	Gestation (g/kg)	Lactation (g/kg)
Wheat		549.0	540.0	0.0	439.3
Canola meal		16.0	75.0	29.0	60.0
Wheat middlings		0.0	10.0	143.0	11.0
Corn DDGS		15.0	25.0	0.0	14.0
Barley		0.0	55.0	580.5	83.0
Limestone		7.0	6.0	20.0	9.0
Soybean meal high protein		205.0	88.0	0.0	138.0
Potash salt		5.5	5.5	5.6	5.9
Lysine		5.0	5.0	0.0	0.0
Methionine		3.0	2.4	0.0	0.0
Animal-Vegetable fat (mix)		37.0	20.0	5.0	13.1
Additives		29.8	8.1	10.0	11.8
Herring fishmeal		1.9	0.0	0.0	0.0
Meat meal		27.5	48.0	0.0	40.0
Whey		60.2	0.0	0.0	0.0
Corn		23.0	37.0	154.0	100.0
Peas		15.0	75.0	53.0	75.0
Nutrients	**Units**	**Nursery 1**	**Nursery 2**	**Gestation**	**Lactation**
Dry matter	%	90.2	89.9	88.8	89.8
Protein	%	20.7	21.1	12.5	19.9
Total P	%	0.62	0.58	0.55	0.58
Total K	%	0.85	0.65	0.59	0.80
Gross Energy	MJ/kg	18.5	17.9	16.5	18.3
Digestible Energy	MJ/kg	16.1	15.6	14.4	15.9
Metabolisable Energy	MJ/kg	14.5	14.0	12.9	14.4

Appendix A2: Composition of experimental diets in Chapter 4

Table A2.5 Ingredients and nutritional composition of the control diet in experiment 1 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated

Ingredient	**Starter (g/kg)**	**Grower (g/kg)**	**Finisher (g/kg)**	**Late finisher (g/kg)**
Canola Meal	104.7	163.7	165.9	204.7
Corn	675.1	694.7	745.6	766.4
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	0.0	0.0	0.0	0.0
Bakery Meal	0.0	0.0	0.0	0.0
Soybean meal de-hulled	169.5	105.0	68.5	11.7
Wheat shorts	0.0	0.0	0.0	0.0
Limestone	12.9	13.2	12.0	10.9
Mono-calcium Phosphate	6.6	5.0	2.6	0.7
Lysine HCL	3.2	2.9	1.7	1.9
DL methionine	0.5	0.1	0.0	0.0
L Threonine	0.9	0.8	0.3	0.3
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	19.1	9.8	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	15.65	14.35	12.87	11.48
Dig Arginine	1.06	0.94	0.82	0.69
Dig Histidine	0.50	0.45	0.40	0.34
Dig Ileum	0.63	0.56	0.50	0.43

Dig Leucine	1.37	1.28	1.19	1.08
Dig Lysine	1.04	0.92	0.73	0.65
Dig Methionine	0.32	0.27	0.25	0.24
Dig Phenylalanine	0.74	0.67	0.60	0.52
Dig Threonine	0.63	0.58	0.49	0.44
Dig Tryptophan	0.17	0.16	0.13	0.11
Dig Valine	0.73	0.67	0.61	0.54
Dig Cysteine	0.27	0.27	0.26	0.25
Dig Meth + Cys	0.59	0.55	0.51	0.49
Ca	0.76	0.76	0.67	0.59
P	0.56	0.54	0.48	0.44
Dig P	0.31	0.28	0.23	0.19
K	0.67	0.62	0.56	0.50
Crude Protein	18.81	17.67	16.04	14.67

Table A2.6 Ingredients and nutritional composition of the meat meal diet in experiment 1 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	97.2	140.3	156.0	183.2
Corn	663.4	683.6	709.5	733.6
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	50.0	50.0	75.0	75.0
Bakery Meal	0.0	0.0	0.0	0.0
Soybean meal de-hulled	153.0	104.1	54.8	4.4
Wheat shorts	0.0	0.0	0.0	0.0
Limestone	7.6	7.4	1.1	0.0
Mono-calcium Phosphate	0.0	0.0	0.0	0.0
Lysine HCL	2.5	1.9	0.3	0.5
DL methionine	0.3	0.0	0.0	0.0
L Threonine	0.7	0.4	0.0	0.0
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	18.0	7.5	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	16.51	15.47	14.58	13.18
Dig Arginine	1.14	1.04	0.97	0.84
Dig Histidine	0.51	0.46	0.42	0.36
Dig Ileum	0.65	0.60	0.56	0.48
Dig Leucine	1.43	1.36	1.30	1.20
Dig Lysine	1.04	0.92	0.73	0.65
Dig Methionine	0.32	0.29	0.29	0.27
Dig Phenylalanine	0.77	0.71	0.66	0.58

Dig Threonine	0.63	0.58	0.51	0.46
Dig Tryptophan	0.17	0.16	0.14	0.12
Dig Valine	0.77	0.73	0.70	0.63
Dig Cysteine	0.27	0.27	0.26	0.25
Dig Meth + Cys	0.59	0.56	0.55	0.53
Ca	0.76	0.76	0.67	0.62
P	0.57	0.59	0.66	0.66
Dig P	0.32	0.33	0.40	0.39
K	0.66	0.62	0.56	0.50
Crude Protein	20.26	19.32	18.69	17.24

Table A2.7 Ingredients and nutritional composition of the bakery meal diet in experiment 1 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	**Starter (g/kg)**	**Grower (g/kg)**	**Finisher (g/kg)**	**Late finisher (g/kg)**
Canola Meal	100.5	158.8	173.4	212.2
Corn	630.5	627.5	648.4	669.3
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	0.0	0.0	0.0	0.0
Bakery Meal	50.0	75.0	100.0	100.0
Soybean meal de-hulled	169.1	103.0	58.6	1.8
Wheat shorts	0.0	0.0	0.0	0.0
Limestone	12.9	13.2	11.8	10.7
Mono-calcium Phosphate	6.4	4.7	2.1	0.1
Lysine HCL	3.3	3.0	1.9	2.1
DL methionine	0.5	0.1	0.0	0.0
L Threonine	1.0	0.8	0.4	0.4
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	18.4	9.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	15.63	14.31	12.82	11.43
Dig Arginine	1.06	0.93	0.81	0.68
Dig Histidine	0.50	0.44	0.39	0.33
Dig Ileum	0.63	0.56	0.50	0.43
Dig Leucine	1.36	1.26	1.16	1.05

Dig Lysine	1.04	0.92	0.73	0.65
Dig Methionine	0.32	0.27	0.25	0.24
Dig Phenylalanine	0.74	0.67	0.60	0.51
Dig Threonine	0.63	0.58	0.49	0.44
Dig Tryptophan	0.18	0.16	0.14	0.11
Dig Valine	0.73	0.67	0.61	0.54
Dig Cysteine	0.27	0.27	0.26	0.25
Dig Meth + Cys	0.59	0.55	0.51	0.50
Ca	0.76	0.76	0.67	0.59
P	0.56	0.54	0.48	0.44
Dig P	0.31	0.28	0.23	0.19
K	0.67	0.62	0.56	0.50
Crude Protein	18.84	17.69	16.15	14.78

Table A2.8 Ingredients and nutritional composition of the corn DDGS diet in experiment 1 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	18.4	58.1	62.5	81.6
Corn	545.1	495.9	554.6	668.7
Corn DDGS	200.0	300.0	300.0	200.0
Meat meal	0.0	0.0	0.0	0.0
Bakery Meal	0.0	0.0	0.0	0.0
Soybean meal de-hulled	169.7	82.4	34.4	16.1
Wheat shorts	0.0	0.0	0.0	0.0
Limestone	14.4	15.4	14.3	12.7
Mono-calcium Phosphate	5.1	2.4	0.0	0.0
Lysine HCL	4.0	4.4	3.6	3.2
DL methionine	0.3	0.0	0.0	0.0
L Threonine	0.7	0.5	0.3	0.4
L Tryptophan	0.0	0.1	0.1	0.1
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	34.9	35.9	26.9	13.9
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	16.45	15.22	13.38	11.75
Dig Arginine	1.05	0.88	0.72	0.63
Dig Histidine	0.52	0.45	0.39	0.34
Dig Ileum	0.66	0.59	0.50	0.43
Dig Leucine	1.63	1.62	1.50	1.31
Dig Lysine	1.04	0.92	0.73	0.65

Dig Methionine	0.32	0.30	0.28	0.25
Dig Phenylalanine	0.82	0.75	0.66	0.57
Dig Threonine	0.63	0.58	0.49	0.44
Dig Tryptophan	0.17	0.15	0.12	0.11
Dig Valine	0.78	0.72	0.64	0.56
Dig Cysteine	0.27	0.27	0.25	0.23
Dig Meth + Cys	0.59	0.57	0.53	0.48
Ca	0.76	0.76	0.67	0.59
P	0.55	0.53	0.46	0.42
Dig P	0.33	0.31	0.25	0.22
K	0.75	0.72	0.64	0.55
Crude Protein	16.48	18.66	16.55	17.52

Table A2.9 Ingredients and nutritional composition of the wheat shorts diet in experiment 1 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	66.7	86.6	24.5	97.8
Corn	498.5	435.5	408.8	626.5
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	0.0	0.0	0.0	0.0
Bakery Meal	0.0	0.0	0.0	0.0
Soybean meal de-hulled	166.2	118.7	115.5	43.6
Wheat shorts	200.0	300.0	400.0	200.0
Limestone	14.6	15.8	14.3	12.3
Mono-calcium Phosphate	3.0	0.0	0.0	0.0
Lysine HCL	3.2	2.5	1.0	1.9
DL methionine	0.6	0.2	0.2	0.0
L Threonine	1.0	0.7	0.1	0.4
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	35.1	32.3	14.1
Animal-vegetable fat blend	38.8	0.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	15.63	14.62	13.49	11.39
Dig Arginine	1.11	1.04	1.00	0.75
Dig Histidine	0.50	0.46	0.44	0.35
Dig Ileum	0.63	0.58	0.54	0.43
Dig Leucine	1.31	1.21	1.14	1.04
Dig Lysine	1.04	0.92	0.73	0.65

Dig Methionine	0.32	0.27	0.25	0.22
Dig Phenylalanine	0.74	0.69	0.66	0.53
Dig Threonine	0.63	0.58	0.49	0.44
Dig Tryptophan	0.19	0.18	0.17	0.12
Dig Valine	0.74	0.71	0.67	0.55
Dig Cysteine	0.27	0.27	0.25	0.23
Dig Meth + Cys	0.59	0.55	0.51	0.45
Ca	0.76	0.76	0.67	0.59
P	0.58	0.58	0.60	0.49
Dig P	0.31	0.29	0.31	0.23
K	0.79	0.81	0.83	0.61
Crude Protein	19.57	19.04	17.97	14.99

Table A2.10 Ingredients and nutritional composition of the OF diet in experiment 2 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	66.7	86.6	24.5	97.8
Corn	498.5	435.5	408.8	626.5
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	0.0	0.0	0.0	0.0
Bakery Meal	0.0	0.0	0.0	0.0
Soybean meal de-hulled	166.2	118.7	115.5	43.6
Wheat shorts	200.0	300.0	400.0	200.0
Limestone	14.6	15.8	14.3	12.3
Mono-calcium Phosphate	3.0	0.0	0.0	0.0
Lysine HCL	3.2	2.5	1.0	1.9
DL methionine	0.6	0.2	0.2	0.0
L Threonine	1.0	0.7	0.1	0.4
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	35.1	32.3	14.1
Animal-vegetable fat blend	38.8	0.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	10.21	9.89	9.72	9.65
Dig Crude Protein	15.63	14.05	12.52	11.05
Dig Arginine	1.06	0.91	0.79	0.67
Dig Histidine	0.49	0.43	0.38	0.32
Dig Ileum	0.63	0.55	0.48	0.41
Dig Leucine	1.36	1.22	1.14	1.04
Dig Lysine	1.04	0.92	0.73	0.65
Dig Methionine	0.32	0.27	0.25	0.22

Dig Phenylalanine	0.74	0.65	0.58	0.51
Dig Threonine	0.63	0.58	0.49	0.44
Dig Tryptophan	0.17	0.15	0.13	0.11
Dig Valine	0.73	0.66	0.59	0.52
Dig Cysteine	0.27	0.27	0.25	0.23
Dig Meth + Cys	0.59	0.55	0.51	0.46
Ca	0.76	0.76	0.67	0.59
P	0.56	0.54	0.48	0.43
Dig P	0.31	0.28	0.23	0.19
K	0.67	0.62	0.56	0.51
Crude Protein	18.91	17.57	15.81	14.16

Table A2.11 Ingredients and nutritional composition of the 0.975 OF diet in experiment 2 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	97.2	159.0	132.1	111.0
Corn	650.0	634.8	674.4	687.0
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	9.7	0.9	0.0	0.0
Bakery Meal	50.0	75.0	0.0	0.0
Soybean meal de-hulled	149.1	81.2	51.6	24.5
Wheat shorts	8.6	23.0	122.4	159.3
Limestone	11.8	13.0	13.0	12.0
Mono-calcium Phosphate	4.6	3.9	0.5	0.0
Lysine HCL	3.4	3.3	2.2	2.3
DL methionine	0.5	0.1	0.1	0.0
L Threonine	1.0	0.9	0.5	0.6
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	6.9	0.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	9.95	9.65	9.48	9.41
Dig Crude Protein	15.24	13.70	12.18	10.84
Dig Arginine	1.02	0.88	0.79	0.68
Dig Histidine	0.48	0.42	0.37	0.32
Dig Ileum	0.61	0.53	0.46	0.40
Dig Leucine	1.34	1.21	1.11	1.01
Dig Lysine	1.01	0.89	0.72	0.63

Dig Methionine	0.31	0.27	0.25	0.22
Dig Phenylalanine	0.72	0.63	0.56	0.50
Dig Threonine	0.62	0.56	0.47	0.43
Dig Tryptophan	0.17	0.15	0.13	0.11
Dig Valine	0.71	0.64	0.58	0.52
Dig Cysteine	0.27	0.27	0.25	0.23
Dig Meth + Cys	0.58	0.53	0.49	0.44
Ca	0.74	0.74	0.65	0.58
P	0.54	0.53	0.48	0.47
Dig P	0.30	0.27	0.22	0.21
K	0.65	0.60	0.60	0.57
Crude Protein	18.52	17.12	15.67	14.24

Table A2.12 Ingredients and nutritional composition of the 0.95 OF diet in experiment 2 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	Starter (g/kg)	Grower (g/kg)	Finisher (g/kg)	Late finisher (g/kg)
Canola Meal	78.3	126.6	70.7	90.8
Corn	625.0	549.6	575.2	644.9
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	14.6	7.5	0.0	0.0
Bakery Meal	50.0	75.0	0.0	0.0
Soybean meal de-hulled	135.8	66.8	66.3	28.7
Wheat shorts	69.9	151.4	268.6	200.0
Limestone	11.6	13.3	13.4	29.6
Mono-calcium Phosphate	2.5	0.4	0.0	0.0
Lysine HCL	3.5	3.4	1.9	2.2
DL methionine	0.5	0.1	0.1	0.0
L Threonine	1.0	0.9	0.4	0.5
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	0.0	0.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	9.70	9.40	9.24	9.16
Dig Crude Protein	14.85	13.35	12.16	10.65
Dig Arginine	1.00	0.88	0.83	0.68
Dig Histidine	0.46	0.40	0.38	0.32
Dig Ileum	0.59	0.51	0.47	0.40
Dig Leucine	1.29	1.15	1.08	0.98
Dig Lysine	0.99	0.87	0.70	0.62

Dig Methionine	0.30	0.26	0.24	0.21
Dig Phenylalanine	0.70	0.61	0.58	0.50
Dig Threonine	0.60	0.55	0.46	0.42
Dig Tryptophan	0.16	0.15	0.14	0.11
Dig Valine	0.69	0.63	0.59	0.51
Dig Cysteine	0.26	0.26	0.24	0.22
Dig Meth + Cys	0.56	0.52	0.48	0.43
Ca	0.72	0.72	0.63	1.25
P	0.54	0.53	0.53	0.48
Dig P	0.29	0.27	0.26	0.22
K	0.66	0.65	0.69	0.59
Crude Protein	18.33	17.25	16.06	14.11

Table A2.13 Ingredients and nutritional composition of the 0.925 OF diet in experiment 2 All ingredient inclusions shown in g/kg as fed, all nutrient levels shown as % as fed unless otherwise stated.

Ingredient	**Starter (g/kg)**	**Grower (g/kg)**	**Finisher (g/kg)**	**Late finisher (g/kg)**
Canola Meal	48.7	81.2	15.1	86.4
Corn	537.1	531.7	484.3	625.1
Corn DDGS	0.0	0.0	0.0	0.0
Meat meal	13.8	0.0	0.0	0.0
Bakery Meal	50.0	0.0	0.0	0.0
Soybean meal de-hulled	125.5	77.4	79.4	28.5
Wheat shorts	200.0	285.7	400.0	200.0
Limestone	12.6	14.9	15.8	54.2
Mono-calcium Phosphate	0.0	0.0	0.0	0.0
Lysine HCL	3.5	3.2	1.6	2.1
DL methionine	0.5	0.2	0.2	0.0
L Threonine	1.0	0.8	0.3	0.5
L Tryptophan	0.0	0.0	0.0	0.0
Canola Oil	0.0	0.0	0.0	0.0
Animal-vegetable fat blend	0.0	0.0	0.0	0.0
Additives	7.4	4.9	3.3	3.3

Resource				
Net Energy (MJ/kg)	9.44	9.15	8.99	8.92
Dig Crude Protein	14.46	13.08	12.12	10.38
Dig Arginine	1.00	0.89	0.87	0.67
Dig Histidine	0.45	0.40	0.38	0.31
Dig Ileum	0.57	0.50	0.47	0.39
Dig Leucine	1.23	1.12	1.05	0.96
Dig Lysine	0.96	0.85	0.68	0.60
Dig Methionine	0.29	0.25	0.23	0.21
Dig Phenylalanine	0.68	0.61	0.59	0.48

Dig Threonine	0.59	0.53	0.45	0.41
Dig Tryptophan	0.16	0.15	0.15	0.11
Dig Valine	0.68	0.63	0.60	0.50
Dig Cysteine	0.26	0.25	0.23	0.22
Dig Meth + Cys	0.55	0.51	0.47	0.42
Ca	0.70	0.70	0.71	2.20
P	0.54	0.55	0.58	0.47
Dig P	0.28	0.27	0.30	0.22
K	0.72	0.73	0.77	0.58
Crude Protein	18.37	17.21	16.37	13.76

Appendix B: Minor ingredient data sources

Table B1 Minor ingredients LCI data sources

	Assumptions	Data sources
Whey		(Nemecek and Kagi, 2007)
Limestone		(Nemecek and Kagi, 2007)
Lysine		(Mosnier et al., 2011b)
Methionine		(Mosnier et al., 2011b)
Herring Fishmeal		(Pelletier, 2006)
Potash salt		(Nemecek and Kagi, 2007)
Meat Meal		(Ramirez et al., 2012)
Animal fat		(Ramirez et al., 2012)
Animal-Vegetable fat (mix)	30% Soybean Oil, 30% Canola Oil, 40% Animal Fat	Expert advice Trouw Nutrition
Peas		(Nemecek and Kagi, 2007)
Additives	Impacts modelled as 30% Lysine, 20% Methionine 50% salt	Expert advice Trouw Nutrition
NaCl		(Nemecek and Kagi, 2007)

Appendix C: On farm energy use data

Table C1 Assumptions of direct energy inputs per pig in LCA in Eastern and Western pig systems adapted from Lammers et al. (2010)

Stage	Electricity (MJ)		Diesel (MJ)		LPG (MJ)	
	East	West	East	West	East	West
Breeding	41.0	41.0	5.1	5.1	52.4	73.4
Nursery	4.0	4.0	2.4	2.4	10.7	14.9
Grower/Finisher	21.0	21.0	11.7	11.7	67.2	94.0

All values in Table C1 were +/- 20% in the model due to the variability of on farm energy use

Appendix D: Description of the manure model

Principles

All NPK not retained by the animal were considered to be excreted in urine or feces. Losses of P and K were considered to be negligible during storage both initially in housing and for all longer term storage methods. Manure was assumed to be left in house for an average period of 7 days in between excretion and movement to storage. Two applications of manure were assumed annually one in spring and one in autumn, thus the average storage time assumed was 3 months. Regional temperatures for May and October were used to represent approximate conditions for manure application. Average temperatures were < 0C for both regions all months between October and April, emissions from outdoor manure storage during these months were assumed to be negligible. Values and ranges for emission factors emission factors for Eastern and Western can be found later in Table D3.

Methane emissions

Methane emissions were considered to occur during housing (enteric) and manure storage. No net CH_4 is assumed to be emitted during manure application to land

Housing emissions

Enteric CH_4 emissions were calculated using the tier 2 methodology shown in equation D1 (Intergovernmental Panel on Climate Change, 2006). CH4 emissions from manure during housing were considered to be negligible.

Equation D1: EF= (GE * (Ym/100) * 365)/55.65

EF = emission factor, kg CH4 per pig

GE = gross energy intake, MJ per pig

Ym = methane conversion factor, % of gross energy in feed converted to enteric methane

The factor 55.65 (MJ/kg CH4) is the energy content of methane

(Ym = 1% sows, 0.39% Growers (Jørgensen et al., 2011))

Storage emissions

Storage CH_4 emissions were equation D2 (Intergovernmental Panel on Climate Change, 2006).

Equation D2: $EF= VS * B_0 * 0.67kg/m^3 * MCF_{S,k} * MS_{S,k}$

EF = emission factor, kg CH_4 per pig

VS = volatile solid excreted per pig

B_0 = maximum methane producing capacity for manure type

0.67 = m^3 to kg conversion of CH_4

MCF $_{(S,k)}$ = methane conversion factor for storage system *S* and climate conditions *k*

MS $_{(S,k)}$ = fraction of manure handled using system *S* in climate *k*

Where Volatile Solids excreted were calculated using equation D3 (Intergovernmental Panel on Climate Change, 2006)

MCF's for storage types and their variation can be found in the parameters list in appendix E

Equation D3: $VS= (GE * (1-DE) * (UE*GE)*(1-ASH/18.45))$

VS = volatile solid excretion per pig, kg VS

GE = gross energy intake, MJ per pig

DE = digestibility of the feed in percent

UE = urinary energy expressed as fraction of GE (assumed to be 0.02)

ASH = the ash content of feed

18.45 = approximate conversion factor for dietary GE per kg of dry matter (MJ kg-1).

Table D1 assumptions regarding storage type (Sheppard et al., 2010b; Statistics-Canada, 2003)

Storage type frequency	East	West
Open tank	0.35	0.55
closed tank	0.24	0.21
Pit below barn	0.21	0.21
anaerobic lagoon	0.14	0.25
Solid (bedding)	0.06	0.03

Nitrogen emissions

The amount of Nitrogen applied to land when after storage was modelled as in equation D4

Equation D4: Napp= Nex-NlossH-NlossS

Napp= N application to soil per pig (kg)

Nex = N excreted per pig (kg)

NlossH = Nitrogen Loss during period of manure storage in housing (kg)

NlossS = Nitrogen loss during storage (kg)

Where N losses during housing calculated as in equation D5

*Equation D5: NLossH = (Nex*EF_NH3_H)+ (Nex*EF_N2O_H)+ (Nex*EF_NOx_H)+ (Nex*EF_N2_H)*

Nex = N excreted per pig (kg)

NlossH = Nitrogen Loss during period of manure storage in housing (kg)

EF_NH3_H = kg N lost as NH3 per kg N excreted as TAN

EF_N2O_H = kg N lost as N2O per kg N excreted

EF_NOx_H = kg N lost as NOx per kg N excreted

EF_N2_H = kg N lost as N2 per kg N excreted

EF_NH3_H was calculated using the information in Table D2 taken from (Sheppard et al., 2010b) – barn temperature was assumed to be on average 2 °C lower in winter than summer. TAN content of manure N was assumed to stabilise within a few hours of excretion after hydrolysis of urea to ammoniacal N had stabilized (Sheppard et al., 2010b). TAN mean value was 70% N excreted with a range of 0.62-0.79

Table D2 Emission factors for NH3 (EF NH3 H) for different floor types during housing.

Floor type	EF_NH3_H Summer	EF_NH3_H Winter	Fraction of floors East	Fraction of floors West
Solid litter	0.21	0.19	0.01	0.03
Solid no litter	0.21	0.19	0.02	0.03
Slurry solid floor	0.31	0.29	0.04	0.01
Part slatted	0.26	0.24	0.47	0.30
Full slatted	0.36	0.34	0.46	0.63
EF_NH3_H	EAST	0.297	WEST	0.309

N_2O emissions during housing were considered to be negligible over the time scale, small NOx and N_2 losses were accounted for see appendix 5 for the emissions factors.

*Equation D6: NLossS = (Ns*EF_NH3_S)+ (Ns*EF_N2O_S)+ (Ns*EF_NOx_S)+ (Ns*EF_N2_S)+(Ns*EF_NO3_S)*

Ns= Nex - NlossH

EF_NH3_S = kg N lost as NH3 per kg Ns

EF_N2O_S = kg N lost as N2O per kg Ns

EF_NOx_S = kg N lost as NOx per kg Ns

EF_N2_S = kg N lost as N2 per kg Ns

EF_NO3_S = kg N lost as NO3 per kg Ns

Where manure stored as slurry

Equation D7: EF_NH3_Sl = 0.13(1-0.058*(15-T))*

EF_NH3_Sl = kg N lost as NH3 per kg Ns (slurry)

T = average temperature over during storage period

Where manure stored as solid manure

Equation D8: EF_NH3_So = 0.13(1-0.058*(17-T))*

EF_NH3_So = kg N lost as NH3 per kg Ns (Solid)

T = average temperature over during storage period

EF_NH3_Sl was reduced by a factor of 4 in cases where a crust cover was used. This prevalence of crust covers was assumed to be 35% in Eastern provinces and 55% in Western (Sheppard et al., 2010b)

Manure Application

The Nitrogen in manure as applied to land was assumed to replace the need to supply approximately 0.75 equivalent N from inorganic fertilizer (Nguyen et al., 2011). The machinery and fuel required in application was assumed to be roughly equal. Therefore the emissions resulting from manure application were calculated as in Equation D9.

*Equation D9: N_Loss_App = N_loss_app_M – (0.75*N_loss_app_s)*

N_Loss_App = net N emissions

N_loss_app_M = N emissions from manure application

N_loss_app_s = N emissions from inorganic fertilizer application

*Equation D10: N_Loss_app_M = (Napp*EFm_NH3_app_) + (Napp*EFm_N2O_app) + (Napp*EFm_NOx_app)) + (Napp*EFm_NO3_app)*

EFm_NH3_app = kg N lost as NH3 per kg N applied in manure

EFm_N2O_app = kg N lost as N2O per kg N applied in manure

EFm_NOx_app = kg N lost as NOx per kg N applied in manure

EFm_NO3_app = kg N lost as NO3 per kg N applied in manure

*Equation D11: N_Loss_app_s = (Napp*EFs_NH3_app) + (Napp*EFs_N2O_app) + (Napp*EFs_NOx_app) + (Napp*EFs_NO3_app)*

EFs_NH3_app = kg N lost as NH3 per kg N applied as inorganic fertilizer

EFs_N2O_app = kg N lost as N2O per kg N applied as inorganic fertilizer

EFs_NOx_app = kg N lost as NOx per kg N applied as inorganic fertilizer

EFs_NO3_app = kg N lost as NO3 per kg N applied as inorganic fertilizer

At all stages indirect N_2O formation was assumed to occur at a rate of 0.01 (NH_3+NOx) and 0.0075 NO_3 (Intergovernmental Panel on Climate Change, 2006), variability in this was modelled (see Table D3 for ranges of all parameters).

The increased emissions of NH3 and N2O account for most (~19% - see emission factors Table D3) of the extra 25% N losses when applying organic manure in the model in comparison to applying mineral fertilizer. The remaining N is assumed to be either emitted as gaseous N_2 or retained as organic N in the soil.

Phosphorus emissions

The net P emissions from PO_4 leaching during application were calculated using the same methodology as those above for NO_3 in manure. The overall likelihood of leaching events was considered to be equal for the two forms of P application and much more dependent on climatic and soil conditions than fertilizer type. The possibility of up to 4% net increase in P leaching was however included in the LCA (see Table D3).

Emission Factors

Table D3 The emission factor in manure model for Eastern and Western Canada for each factor the input mean, maximum (max) and minimum (min) is shown. In the case of normally distributed parameters the max and min values shown here represent the upper and lower 95% confidence intervals of their distribution

		Eastern Canada			Western Canada			
Emission Factor	Definitions	Mean	Min	Max	Mean	Min	Max	Sources
Bo	Maximum m^3 CH_4 per kg VS excreted	0.48	0.43	0.53	0.48	0.43	0.53	(Intergovernmental Panel on Climate Change, 2006)
EF_NH3_H	Kg NH3-N emitted/kg TAN excreted housing	0.297	0.247	0.347	0.309	0.259	0.359	(Sheppard et al., 2010b)
EF_NOx_H	Kg NOx-N emitted / kg N excreted housing	0.002	0.0015	0.0025	0.002	0.0015	0.0025	(Nguyen et al., 2011)
EF_NOx_S	Kg NOx-N emitted / kg N stored	0.005	0.004	0.006	0.005	0.004	0.006	(Nguyen et al., 2012)
EF3_N2O_AL	Kg N2O-N emitted / kg N stored in Anaerobic Lagoon	0.0035	0.0025	0.035	0.0035	0.0025	0.035	(Liu et al., 2013)
EF3_ N2O _CT	Kg N2O-N emitted / kg N stored in concrete tank solid cover	0	0	0.0001	0	0	0.0001	(Liu et al., 2013)
EF3_ N2O _OT	Kg N2O-N emitted / kg N stored in concrete tank open	0.0001	0	0.0002	0.0001	0	0.0002	(Liu et al., 2013)
EF3_ N2O _Pit	Kg N2O-N emitted / kg N stored in slurry stored below barn	0.0006	0	0.0019	0.0006	0	0.0019	(Liu et al., 2013)

EF3_ N2O _SB	Kg N2O-N emitted / kg N stored as solid manure	0.0002	0	0.0004	0.0002	0	0.0004	(Liu et al., 2013)
EF_N2_H	Kg N2-N emitted / kg N excreted Housing	0.002	0.0015	0.0025	0.002	0.0015	0.0025	(Nguyen et al., 2011)
EF_N2_S	Kg N2-N emitted / kg N stored	0.015	0.012	0.018	0.015	0.012	0.018	(Nguyen et al., 2011)
EFm_N2O_app	Kg N2O-N emitted / kg N applied to land manure	0.0204	0.0104	0.0304	0.006	0	0.016	(Bouwman et al., 2002; Rochette et al., 2008)
EFm_NH3_app	Kg NH3-N emitted / kg N applied to land manure	0.257	0.2313	0.2827	0.198	0.178	0.218	(Sheppard et al., 2010b)
EFm_NO3_app	Kg NH3-N leached / kg N applied to land manure	0.2	0.05	0.3	0.1	0.05	0.3	(Intergovernmental Panel on Climate Change, 2006; Rochette et al., 2008)
EFm_NOX_app	Kg NOx-N leached / kg N applied to land manure	0.001	0	0.002	0.001	0	0.002	(Nguyen et al., 2011)
EFm_PO4_app	Kg PO4-P emitted / kg P applied to land manure	0.02	0	0.04	0.02	0	0.04	(Nguyen et al., 2011)
EFs_N2O_app	Kg N2O-N emitted / kg N applied to land inorganic fertilizer	0.017	0.0111	0.0229	0.005	0.0026	0.0074	(Bouwman et al., 2002; Rochette et al., 2008)
EFs_NH3_app	Kg NH3-N emitted / kg N applied to land inorganic fertilizer	0.079	0.065	0.09	0.055	0.045	0.063	(Sheppard et al., 2010a)

EFs_NO3_app	Kg NO3-N emitted / kg N applied to land inorganic fertilizer	0.2	0.05	0.3	0.1	0.05	0.3	(Intergovernmental Panel on Climate Change, 2006; Rochette et al., 2008)
EFs_NOX_app	Kg NOx-N emitted / kg N applied to land inorganic fertilizer	0.007	0	0.014	0.007	0	0.014	(Nguyen et al., 2011)
EFs_PO4_app	Kg PO4-P emitted / kg P applied to land inorganic fertilizer	0.02	0	0.04	0.02	0	0.04	(Nguyen et al., 2011)
MCF_AL	Methane Conversion Factor Anaerobic Lagoon (decimal)	0.44	0.24	0.64	0.44	0.24	0.64	(Liu et al., 2013)
MCF_CT	Methane Conversion Factor closed concrete tank slurry(decimal)	0.1	0.02	0.18	0.1	0.02	0.18	(Liu et al., 2013)
MCF_OT	Methane Conversion Factor closed open tank slurry(decimal)	0.17	0.07	0.27	0.17	0.07	0.27	(Liu et al., 2013)
MCF_Pit	Methane Conversion Factor slurry stored beneath barn(decimal)	0.17	0.07	0.27	0.17	0.07	0.27	(Liu et al., 2013)
MCF_SB	Methane Conversion Factor solid manure storage	0.02	0.01	0.03	0.02	0.01	0.03	(Liu et al., 2013)
N2O_Vol_NH3	Kg N_2O-N formed / kg NH_3-N+ NO_X-N volatized	0.01	0.005	0.015	0.01	0.005	0.015	(Intergovernmental Panel on Climate Change, 2006)

NO3_lag	Kg NO3-N leached / kg N stored in unlined lagoon	0.2	0.1	0.4	0.2	0.1	0.4	(Prapaspongsa et al., 2010)
N2O_vol_NO3	Kg N_2O-N formed / kg NO3 leached	0.0075	0.00375	0.01125	0.0075	0.00375	0.01125	(Intergovernmental Panel on Climate Change, 2006)
K_replace_rate	Replacement rate of inorganic K by K in manure	1	0.9	1	1	0.9	1	(Nguyen et al., 2011)
N_replace_rate	Replacement rate of inorganic N by N in manure	0.75	0.5	1	0.75	0.5	1	(Nguyen et al., 2011)
P_replace_rate	Replacement rate of inorganic P by P in manure	0.9	0.8	1	0.9	0.8	1	(Nguyen et al., 2011)
T_summer	Average temperature 6 months summer (C)	13.55	11.55	15.55	11.7	9.7	13.7	(Weatherbase, 2014)
TAN	Total Ammomiacal Nitrogen fraction of manure N	0.7	0.62	0.79	0.7	0.62	0.79	(Sheppard et al., 2010b)
Ym_Sows	% gross energy in feed converted to enteric methane sows	0.01	0	0.02	0.01	0	0.02	(Jørgensen et al., 2011))
Ym_Growers	% gross energy in feed converted to enteric methane growers	0.0039	0.00312	0.00468	0.0039	0.00312	0.00468	(Jørgensen et al., 2011))

Appendix E: The mean values and uncertainty ranges of the LCA model parameters for Eastern Canada

Table E1 The mean values and uncertainty ranges of the model parameters for Eastern Canada. In the case of normally and lognormally distributed parameters the max and min values shown here represent the upper and lower 95% confidence intervals of their distribution

Parameters Name	Definitions	Value	Distribution	lower	upper	Sources[1]
FCR_GF_E	Feed conversion Ratio Grower/Finisher	2.74	Triangle	2.5	3.09	(Mackenzie et al., 2015)
FCR_Nurs_E	Feed conversion Ratio Nursery	1.57	Triangle	1.38	1.8	(Mackenzie et al., 2015)
FI_Sow_E	Feed intake per sow	40.6	Triangle	35	49.9	(Mackenzie et al., 2015)
Body_N	kg N / kg Live Weight	0.0256	Normal	0.02432	0.02688	(Wellock et al., 2003)
Body_P	kg P / kg Live Weight	0.005	Normal	0.00475	0.00525	(Symeou et al., 2014)
Body_K	kg K / kg Live Weight	0.002	Normal	0.0019	0.0021	(Lenis and Jongbloed, 1995)
Breeding_N_East	kg N / kg feed breeding	0.0247	Normal	0.02223	0.02717	(Mackenzie et al., 2015)

Breeding_P_East	kg P / kg feed breeding	0.0055	Normal	0.00495	0.00605	(Mackenzie et al., 2015)
Breeding_K_East	kg K / kg feed breeding	0.0067	Normal	0.00603	0.00737	(Mackenzie et al., 2015)
Carcass_yield	Live Weight x Kill Out %	0.8	Normal	0.784	0.816	(Mackenzie et al., 2015; Vergé et al., 2009)
Diesel_conv_GF_E	Diesel input per pig Grower/Finisher (MJ)	6.4	Normal	5.12	7.68	(Lammers et al., 2010)
Diesel_conv_Breed_E	Diesel input per pig breeding (MJ)	5.1	Normal	3.9	6.3	(Lammers et al., 2010)
Diesel_conv_Nurs_E	Diesel input per pig Nursery (MJ)	2.45	Normal		2.94	(Lammers et al., 2010)
Electricity_conv_nursery_E	Electricity input per pig Nursery (MJ)	3.95	Normal	3.16	4.74	(Lammers et al., 2010)
Electricity_conv_Breed_E	Electricity input per pig Breeding (MJ)	41	Normal	32.8	49.2	(Lammers et al., 2010)
Electricity_conv_GF_E	Electricity input per pig Grower/Finisher (MJ)	21	Normal	16.8	25.2	(Lammers et al., 2010)

Finish_E	Final Body Weight (kg live weight)	123.6	Triangle	118	130	(Mackenzie et al., 2015)
LPG_conv_breed_E	LPG input per pig breeding(MJ	52.44	Normal	42.04	62.84	(Lammers et al., 2010)
LPG_conv_GF_E	LPG input per pig Grower/Finisher (MJ	67.2	Normal	53.76	80.64	(Lammers et al., 2010)
LPG_conv_nursery_E	LPG input per pig nursery(MJ	10.68	Normal	8.544	12.816	(Lammers et al., 2010)
Litter_annum_E	Litters/annum sows	2.45	Triangle	2.2	2.55	(Mackenzie et al., 2015)
Mortality_fin_E	Mortality Grower/Finisher phase (decimal)	0.04	Triangle	0.015	0.09	(Mackenzie et al., 2015)
Mortality_nurs_E	Mortality Nursery phase (decimal)	0.028	Triangle	0.0064	0.075	(Mackenzie et al., 2015)
Mortality_sow_E	Mortality Sows per annum (decimal	0.068	Triangle	0.04	0.1	(Mackenzie et al., 2015)
Nursery_GE_East	Gross Energy (MJ) / kg feed Nursery	18.06	Normal	17.699	18.421	(Mackenzie et al., 2015)
Nursery_K_East	kg K / kg feed Nursery	0.0074	Normal	0.00666	0.00814	(Mackenzie et al., 2015)

Nursery_N_East	kg N / kg feed Nursery	0.0313	Normal	0.02817	0.03443	(Mackenzie et al., 2015)
Nursery_P_East	kg P / kg feed Nursery	0.0066	Normal	0.00594	0.00726	(Mackenzie et al., 2015)
Sow_Cull_E	Sows culled per annum (decimal)	0.361	Triangle	0.22	0.58	(Mackenzie et al., 2015)
Start_Nurs_E	Start weight nursery (kg live weight)	6.32	Triangle	5.5	7.3	(Mackenzie et al., 2015)
Weaned_litter_E	Average size of weaned litter	11	Triangle	9.7	11.6	(Mackenzie et al., 2015)
Piglet_mortality_E	Post birth mortality (decimal)	0.125	Triangle	0.062	0.192	(Mackenzie et al., 2015)
Nurs_end_E	End weight nursery (kg live weight)	27.4	Triangle	21	34	(Mackenzie et al., 2015)
Canola_Yield	Kg/hectare Canola	1900	Normal	1600	2200	(Statistics-Canada, 2014a)
Corn_Yield	Kg/hectare Corn	9000	Normal	8000	10000	(Statistics-Canada, 2014a)
Wheat_Yield	Kg/hectare wheat	2800	Normal	2200	3400	(Statistics-Canada, 2014a)
Soybean_yield	Kg/hectare soy	3200	Normal	2600	3800	(Statistics-Canada, 2014a)
Bo	Maximum m^3 CH_4 per	0.48	Normal	0.43	0.53	(Intergovernmental

	kg VS excreted					Panel on Climate Change, 2006)
EF_NH3_E	Kg NH3-N emitted/kg TAN excreted housing	0.297	Normal	0.247	0.347	(Sheppard et al., 2010b)
EF_NOx_H	Kg NOx-N emitted / kg N excreted	0.002	Normal	0.0015	0.0025	(Nguyen et al., 2011)
EF_NOx_S	Kg NOx-N emitted / kg N stored	0.005	Normal	0.004	0.006	(Nguyen et al., 2011)
EF3_AL	Kg N2O-N emitted / kg N stored in Anaerobic Lagoon	0.0035	Triangle	0.0025	0.035	(Liu et al., 2013)
EF3_CT	Kg N2O-N emitted / kg N stored in concrete tank solid cover	0	Undefined	0	0.0001	(Liu et al., 2013)
EF3_OT	Kg N2O-N emitted / kg N stored in concrete tank open	0.0001	Normal	0	0.0002	(Liu et al., 2013)
EF3_Pit	Kg N2O-N emitted / kg N stored in slurry stored below barn	0.0006	Triangle	0	0.0019	(Liu et al., 2013)

EF3_SB	Kg N2O-N emitted / kg N stored as solid manure	0.0002	Normal	0	0.0004	(Liu et al., 2013)
EF_N2_H	Kg N2-N emitted / kg N excreted Housing	0.002	Normal	0.0015	0.0025	(Nguyen et al., 2011)
EF_N2_S	Kg N2-N emitted / kg N stored	0.015	Normal	0.012	0.018	(Nguyen et al., 2011)
EFm_N2O_app_E	Kg N2O-N emitted / kg N applied to land manure	0.0204	Normal	0.0104	0.0304	(Bouwman et al., 2002; Rochette et al., 2008)
EFm_NH3_app_E	Kg NH3-N emitted / kg N applied to land manure	0.257	Normal	0.2313	0.2827	(Sheppard et al., 2010b)
EFm_NO3_app_e	Kg NH3-N leached / kg N applied to land manure	0.2	Undefined	0.05	0.3	(Intergovernmental Panel on Climate Change, 2006; Rochette et al., 2008)
EFm_NOX_app_E	Kg NOx-N leached / kg N applied to land manure	0.001	Normal	0	0.002	(Nguyen et al., 2011)

EFm_PO4_app_e	Kg PO4-P emitted / kg P applied to land manure	0.02	Undefined	0	0.04	(Nguyen et al., 2011)
EFs_N2O_app_E	Kg N2O-N emitted / kg N applied to land inorganic fertilizer	0.017	Normal	0.0111	0.0229	(Bouwman et al., 2002; Rochette et al., 2008)
EFs_NH3_app_E	Kg NH3-N emitted / kg N applied to land inorganic fertilizer	0.079	Triangle	0.065	0.09	(Sheppard et al., 2010a)
EFs_NO3_app_e	Kg NO3-N emitted / kg N applied to land inorganic fertilizer	0.2	Undefined	0.05	0.3	(Intergovernmental Panel on Climate Change, 2006; Rochette et al., 2008)
EFs_NOX_app_E	Kg NOx-N emitted / kg N applied to land inorganic fertilizer	0.007	Normal	0	0.014	(Nguyen et al., 2011)
EFs_PO4_app_e	Kg PO4-P emitted / kg P applied to land inorganic fertilizer	0.02	Undefined	0	0.04	(Nguyen et al., 2011)

MCF_ALE	Methane Conversion Factor Anaerobic Lagoon (decimal)	0.44	Normal	0.24	0.64	(Liu et al., 2013)
MCF_CTE	Methane Conversion Factor closed concrete tank slurry(decimal)	0.1	Normal	0.02	0.18	(Liu et al., 2013)
MCF_OTE	Methane Conversion Factor closed open tank slurry(decimal)	0.17	Normal	0.07	0.27	(Liu et al., 2013)
MCF_PitE	Methane Conversion Factor slurry stored beneath barn(decimal)	0.17	Normal	0.07	0.27	(Liu et al., 2013)
MCF_SBE	Methane Conversion Factor solid manure storage	0.02	Normal	0.01	0.03	(Liu et al., 2013)
N2O_Vol_NH3	Kg N_2O-N formed / kg NH_3-N+ NO_X-N volatized	0.01	Normal	0.005	0.015	(Intergovernmental Panel on Climate Change, 2006)

NO3_lag	Kg NO3-N leached / kg N stored in unlined lagoon	0.2	Triangle	0.1	0.4	(Prapasponga et al., 2010)
NO3_vol	Kg N_2O-N formed / kg NO3 leached	0.0075	Normal	0.00375	0.01125	(Intergovernmental Panel on Climate Change, 2006)
K_replace_rate	Replacement rate of inorganic K by K in manure	1	Triangle	0.9	1	(Nguyen et al., 2011)
N_replace_rate	Replacement rate of inorganic N by N in manure	0.75	Normal	0.5	1	(Nguyen et al., 2011)
P_replace_rate	Replacement rate of inorganic P by P in manure	0.97	Triangle	0.8	1	(Nguyen et al., 2011)
T_summer_E	Average temperature 6 months summer (C)	13.55	Normal	11.55	15.55	(Weatherbase, 2014)
TAN	Total Ammomiacal Nitrogen fraction of manure	0.7	Triangle	0.62	0.79	(Sheppard et al., 2010b)

Ym_Sows	% gross energy in feed converted to enteric methane sows	0.01	Normal	0	0.02	(Jørgensen et al., 2011))
Ym_Growers	% gross energy in feed converted to enteric methane growers	0.0039	Normal	0.00312	0.00468	(Jørgensen et al., 2011))

[1] Where Chapter 3 cited, the parameter mean and range were estimated using benchmark data collected for that study on pig production in Eastern Canada

Appendix F: Price ratios used for diet formulation (Chapter 4)

Table F1 price ratios used for diet formulation in Chapter 4 all prices scaled to the price of yellow dent corn which = 1 per tonne. Data reflected average Canadian (not regionalised) prices for 2013 provided by Trouw Nutrition based on Statistics Canada price data

Ingredient	Price Ratio
Canola Meal	1.6
Corn	1.0
Corn DDGS	1.2
Meat meal (Pork meal)	2.7
Bakery Meal	1.1
Soybean meal dehulled	2.2
Wheat soft	1.0
Wheat Bran	1.9
Wheat DDGS	1.3
Wheat Millrun	0.7
Wheat shorts	0.9
Limestone	0.3
Mono-calcium Phosphate	2.9
Salt	0.3
Lysine HCL	8.1
DL Methionine	17.1
L Threonine	10.6
L Tryptophan	133.8
Soybean Oil	5.8
Canola Oil	4.4
AV fat blend	3.4
Choice white grease	3.4
Additives (fixed inclusion)	20.7

Appendix G: Co-product allocation data

Table G1 Allocation factors used for multioutput processes in the feed supply chain

Multioutput system	By products	Mass yield (%)	Price Ratio [1]	Allocation (%)
Soybean Oil extraction	Soybean meal	77.3	1	43.7
	Soybean Oil	22.7	2.64	56.3
Canola Oil extraction	Canola Meal	57.3	1	32.8
	Canola Oil	42.6	2.76	67.2
Bioethanol production from corn	Ethanol			97.6
	Corn DDGS			2.4
Wheat Flour mill	Flour	73	1[3]	89.8
	Wheat Shorts	12.5	0.22	3.4
	Wheat Bran	12	0.44	6.5
	Wheat Germ	2.0	0.11	0.27
Industrial Bakery [2]	Bread	92	10	99
	Bakery waste	8	1	1
Fat Rendering	Fat	57.7	1.22	62.6
	Meat Meal	42.3	1	37.4

[1] Price data average Canadian (not regionalised) prices for 2013 provided by Trouw Nutrition based on Statistics Canada price data

[2] Expert advice from Sugarich (specialist producers of animal feed using bakery waste products, 2015

[3] Flour price was estimated using the principle that sales of flour provide around 90% of the gross margin for typical wheat flour milling operations (FAO, 2009).

Appendix H: Regional price ratios used for diet formulation (Chapter 5)

Table H1 price ratios used for diet formulation, all prices scaled to the price of wheat which = 1 per tonne. Average ingredient prices and availability in Ontario and Manitoba for 2015 were provided by Trouw Nutrition (derived from Statistics Canada data (Statistics-Canada, 2014b)).

Ingredient	Price Ratio – Eastern Canada	Price Ratio – Western Canada
Barley	0.79	1.01
Bakery meal	1.00	NA
Canola meal	1.46	1.56
Corn	0.75	NA
Corn DDGS	0.98	1.21
Field Peas	N/A	1.17
Meat (pork) meal	2.46	2.88
Soybean meal	1.93	2.43
Wheat	1.00	1.19
Wheat Bran	1.46	1.90
Wheat shorts	0.73	0.89
Animal-vegetable fat blend	3.25	3.43
Canola oil	13.9	NA
Soybean Oil	4.22	4.42
HCL-Lysine	8.17	10.5
L-Threonine	17.7	25.7
FU-Methionine	18.0	30.2
L-Tryptophan	89.3	121
Sodium Chloride	0.31	0.72
Dicalcium Phosphate	2.71	3.39
Limestone	0.44	0.64

Appendix I: Ingredient inclusion limits

Table I3 The maximum inclusion limits (g/kg as fed) of the ingredients for each feeding phase when formulating grower/finisher diets in this study. These limits were based on guidance for pig farmers provided by OMAFRA (OMAFRA, 2012a) as well as peer reviewed studies in the case of some important co-products (see Chapter 4).

Ingredient	Starter	Grower	Finisher	Late finisher
Barley	800	800	800	800
Bakery meal	50	100	100	100
Canola meal	100	100	100	100
Corn	800	800	800	800
Corn DDGS	150	200	200	200
Field Peas	100	100	100	100
Meat (pork) meal	50	50	50	50
Soybean meal	250	250	250	250
Wheat	700	700	700	700
Wheat Bran	50	50	50	50
Wheat shorts	200	300	300	200
Animal-vegetable fat blend[1]	50	50	50	50
Canola oil[1]	20	20	20	20
Soybean Oil[1]	20	20	20	20
HCL-Lysine	10	10	10	10
L-Threonine	10	10	10	10
DL-Methionine	10	10	10	10
L-Tryptophan	10	10	10	10
Sodium Chloride	10	10	10	10
Dicalcium Phosphate	50	50	50	50
Limestone	50	50	50	50

[1] Total fat supplementation was restricted to 50 g/kg as fed in all diets