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How effective can environmental taxes be in reducing the environmental impact of pig farming systems?

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Abstract: Environmental taxes are a form of incentive regulation available to governments to drive reductions in environmental impact. The aims of this study were to: 1) develop a framework that enabled quantification of the potential effect of environmental taxes on pig diet composition and 2) examine the relationship between tax level and its effectiveness in reducing environmental impacts from pig systems. Three taxes were investigated: a carbon tax on the feed ingredients as purchased, and two financial penalties on the field spreading of N and P in manure respectively. Each tax was integrated into a diet-formulation model 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 spatial variation in the consequences of tax measures on diet-formulation. 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, acidification potential, eutrophication potential and non-renewable resource use. As environmental tax levels increased, trigger points in the tax range caused dietary change which reduced levels of the targeted emission type. In almost all the tax scenarios the largest reductions in the target emission per Canadian dollar (C$) increase in cost were achieved at the lower end of the tax range tested, as diminishing marginal returns were evident. The taxes on spreading N and P in manure did not significantly reduce levels of any environmental impact category tested in most 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 considered. These results showed the potential for taxes which target specific emissions, to increase system-

level environmental impacts in livestock production. The study demonstrated how system-level environmental impact models can be used to quantify the potential of environmental taxes set at different rates to reduce overall environmental impact levels in livestock systems.

1. 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 taking decisions concerning environmental taxes in order to reduce the environmental impact of livestock production systems, policy makers need to consider the following issues.

Firstly, 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 recent 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 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.

Secondly, at which point in the production system should taxes be levied in order to be most effective? 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 (Basset-Mens and Van Der Werf, 2005; Leinonen et al., 2012; Williams et al., 2006). For example, feed production contributes around 65% of global warming potential caused by Canadian pig farming systems, and emissions from housing and manure management causing around 70% of eutrophication from Canadian pig farming systems (Mackenzie et al., 2015). Administrative feasibility is 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.

Thirdly 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 formulation of animal diets. Quantitative modelling provides a suitable means to evaluate the implications of adopting different diets in livestock systems for environmental impacts. 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 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). A previous study showed that whilst attempting to reduce the environmental impact of a 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 (Mackenzie et al., 2016a). Therefore it could be expected that 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.

Here we develop a diet-formulation tool designed for pig farming systems in Canada, combined with an LCA model of these systems as an exemplar to investigate the potential implications of environmental taxes on diet-formulation and the environmental impact of livestock systems.

The aims of this study were threefold:

1) to develop a framework that enabled quantification of the potential effects of environmental taxes on pig diet composition;

2) to quantify the implications of dietary alterations caused by environmental taxes for the environmental impacts of the production system using multiple environmental impact categories, and
3) to 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 model which was capable of formulating diets for environmental impact objectives (Mackenzie et al., 2016a): 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. In each case diets were formulated in two scenarios for Eastern and Western Canada respectively. 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. Testing the tax scenarios in two regions allowed any spatial differences in policy implications for environmental impacts and cost to be quantified. It was hypothesised that the diet-formulation model 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.

2. Materials and Methods

2.1 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 phase where pigs are fattened from around 30kg to slaughter weight (PorkCheckoff, 2009). Figure

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 was consumed per pig produced during the grower/finisher phase with at least 75% of the environmental impacts caused by the grower/finisher phase for multiple environmental impact categories (Mackenzie et al., 2015). This study therefore, concentrated on the potential effect of environmental taxes on diets formulated for the grower/finisher phase of production only. The breeding and nursery production stages were treated as independent to the grower/finisher phase in this study and remained constant for all comparisons made.

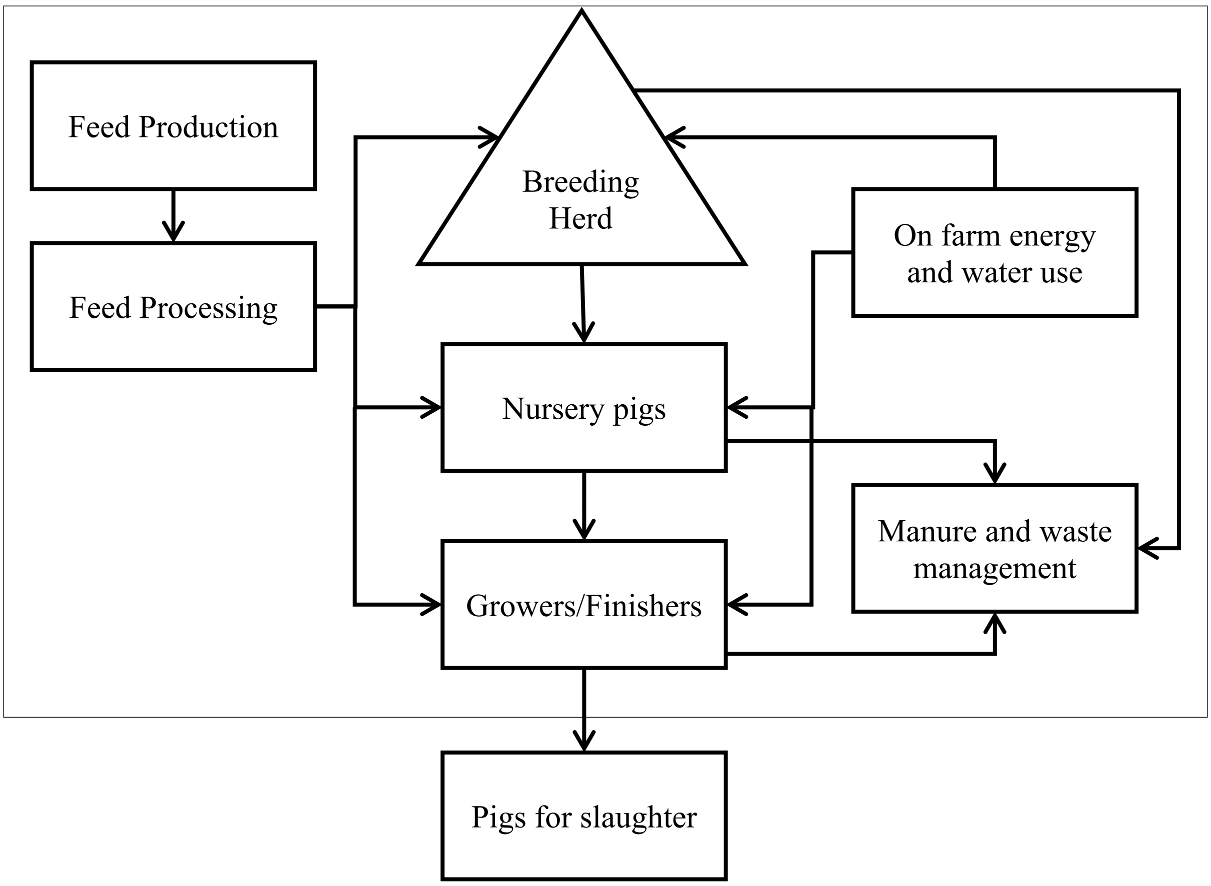


Figure 1 The structure and main components of the pig production system in the Life Cycle Assessment model.

2.2 Diet-formulation

A linear programming algorithm for diet-formulation was used to formulate grower/finisher diets for each taxation scenario; the diet-formulation rules used are described in detail in Mackenzie et al. (2015). In this study all diets were formulated to minimise feed cost per kg live weight gain (least-cost) for the grower/finisher phase in each tax scenario. Explanation on the nutritional rules used to formulate the diets can be found in Appendix 1.

There were 5 broad groups of ingredients used in the diet-formulation model; 1) whole cereals such as wheat and maize, 2) protein meals such as soymeal and canola meal, 3) co-products of other production processes, such as wheat shorts from flour milling and corn dried distillers grains with solubles, 4) specialist ingredients 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 (Mackenzie et al., 2016b). Further explanation of the rules used on ingredient limits can also be found in appendix 1. Average ingredient prices and availability in Ontario and Manitoba for 2015 were provided by TrouwAgresearch, derived from Statistics Canada data (Statistics-Canada, 2014). The price ratios and available ingredients for Eastern and Western Canada can be found in Appendix 2.

The diet-formulation model 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 quantify 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.

2.3 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 Mackenzie et al. 2015 for a 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 in Mackenzie et al. (2016); for details regarding feed ingredients see Appendix 3, for data regarding on farm energy use see Appendix 4 and for details of the manure model see Appendix 5.

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. GWP was quantified in CO_2 equivalents (eq) on a 100 year timescale using the IPPC methodology (IPPC, 2006). The methodology of accounting for GWP caused by land use change in this study followed the PAS 2050 guidelines (BSI, 2011). The impact categories AP (SO_2 eq) and EP (PO_4 eq) 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 system`s contribution to these issues. A fourth impact category, which quantified the Non-Renewable Resource Use (NRRU) of the system 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). NRRU was calculated by aggregating the total non-renewable materials used across the whole system, each material used was weighted in the methodology according to scarcity and units of antimony equivalents were used as the scale (Sb eq). 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 (CML), Leiden University (CML, 2001). 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, 2014). Please refer to Appendix 3 in the supplementary material for further details on how co-product allocation was implemented in the feed supply chain.

2.4 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. This method of uncertainty analysis to distinguish between two scenarios with an LCA model was described in detail in Mackenzie et al. (2015). Further explanation on this methodology and the major causes of uncertainty in the model is in appendix 6. The key output of the simulations was the frequency in which the environmental impact of each tax scenario was greater or smaller than the no -tax 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. The statistical output of the uncertainty analysis in this study tested the hypotheses that each tax scenario caused a reduction in the levels of environmental impact, for one or more of the impact categories tested in the LCA model.

2.5 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 model was a diet composition which minimised feed cost/ kg LW gain during the grower/finisher 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 2.

The carbon tax was added to the price of each ingredient in the diet-formulation model 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 (Mackenzie et al., 2016b, 2015), the GWP values used per kg of each ingredient in the diet-formulation model can be found in Table 1. The effect of a carbon tax on least-cost grower/finisher 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 in an effort to tackle climate change (World Bank, 2013). Moreover many companies (including Google, Disney, Wal-Mart 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).

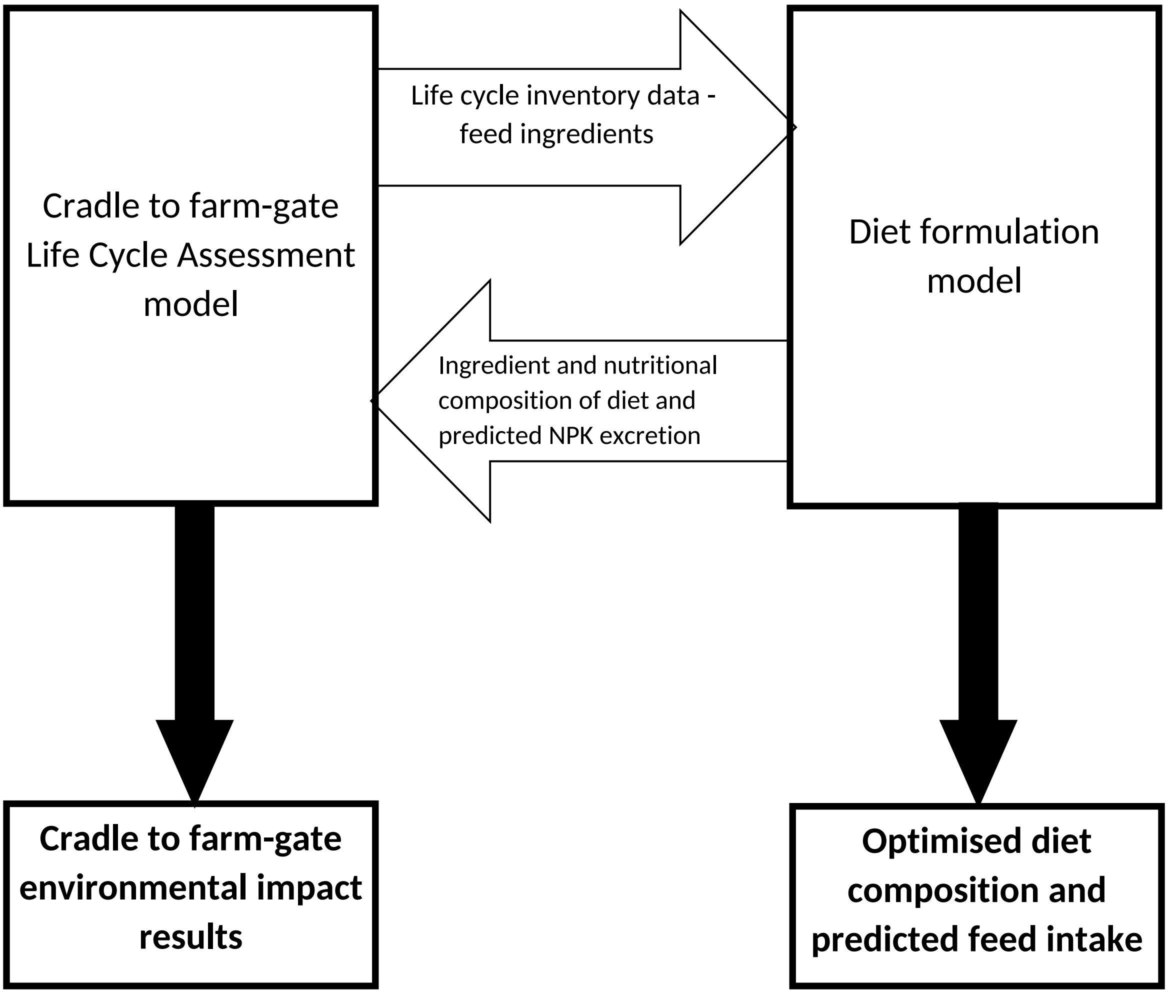


Figure 2 Schematic of the methodology followed to formulate diets in different environmental tax scenarios and then test these diets in the life cycle assessment model to determine the resulting environmental impacts.

Scenarios were modelled for taxes applied to N and P in the grower/finisher 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 1980s (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 25% restriction was used to ensure some consistency when comparing different tax scenarios, as well as to exclude scenarios where taxes made the farming systems economically unviable. 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 model 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 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 a previous life cycle assessment studies of Canadian pig farming systems (Mackenzie et al., 2016b, 2015).

Impact category [1] **Unit** [2]	**NRRU**[3] **kg Sb eq**	**AP**[3] **kg SO2 eq**	**EP**[3] **kg PO4 eq**	**GWP**[3] **kg CO2 eq**
Barley	2.18E-03	5.36E-03	2.69E-03	3.80E-01
Canola meal	1.39E-03	7.97E-03	1.59E-03	3.00E-01
Canola oil	3.84E-03	2.20E-02	4.40E-03	8.40E-01
Maize	1.71E-03	5.13E-03	1.11E-03	3.90E-01
Soya meal	5.70E-04	4.11E-03	8.71E-04	1.50E-01
Wheat	1.84E-03	1.01E-02	2.04E-03	4.30E-01
Meat (pork) meal	1.05E-03	2.46E-04	6.16E-05	1.30E-01
Corn dried distillers grains with solubles	6.51E-03	1.13E-03	2.66E-04	7.80E-01
Wheat Bran	1.02E-03	5.56E-03	1.12E-03	2.40E-01
Wheat shorts	5.12E-04	2.78E-03	5.59E-04	1.20E-01
Field Peas	1.32E-03	2.31E-03	2.72E-03	5.80E-01
Bakery meal	5.17E-04	1.41E-03	2.60E-04	8.00E-02
Animal-vegetable fat blend	2.57E-03	1.01E-02	2.06E-03	4.90E-01
Soy Oil	1.51E-03	1.09E-02	2.30E-03	4.00E-01
HCL-Lysine	3.51E-02	2.12E-02	9.97E-03	4.81E+00
L-Threonine	3.51E-02	2.12E-02	9.97E-03	4.81E+00
FU-Methionine	3.64E-02	7.54E-03	1.70E-03	2.95E+00
L-Tryptophan	7.01E-02	4.24E-02	1.99E-02	9.62E+00
Sodium Chloride	1.21E-03	8.97E-04	6.68E-04	1.80E-01
Dicalcium Phosphate	9.40E-03	2.68E-02	3.63E-04	1.51E+00
Limestone	1.31E-04	1.03E-04	3.58E-05	2.00E-02

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

[2] eq, equivalent

[3] GWP was quantified in CO_2 equivalents (eq) on a 100 year timescale using the IPPC methodology (IPPC, 2006). 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 (CML), Leiden University (CML, 2001).

3. Results and discussion

3.1 Carbon tax

The ingredient compositions of the diets formulated at different levels of carbon taxation are shown in figures 3a and 3b. The relative feed cost, feed intake, N excreted, P excreted, NRRU, AP, EP and GWP for each tax level are shown in figures 4a and 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 in the East. 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 dried distillers grains with solubles included in the diet compared to the no-tax scenario, as corn dried distillers grains with solubles had high levels of GWP per kg associated with it compared to other ingredients accordingly (see Table 1). Secondly at tax levels of C$40 per tonne CO_2 eq and above, soymeal 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 1). 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$60 per tonne CO_2eq and above ($P<0.001$), increasing feed cost by 10%. A carbon tax of C$40 per tonne CO_2eq reduced GWP by 8% at 7% cost increase compared to the no-tax scenario ($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. The amount of soymeal included in the grower/finisher 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 a low tax liability per kg (Table 1). Combined, these two factors contributed to an increase in nutrient excretion as the carbon tax increased. Similar to the Eastern scenario as levels of carbon tax were increased the least-cost diet included less corn dried distillers grains with solubles due to high levels of tax on this ingredient. The nutritional density of the least-cost grower/finisher diets did not change at any level of carbon tax tested.

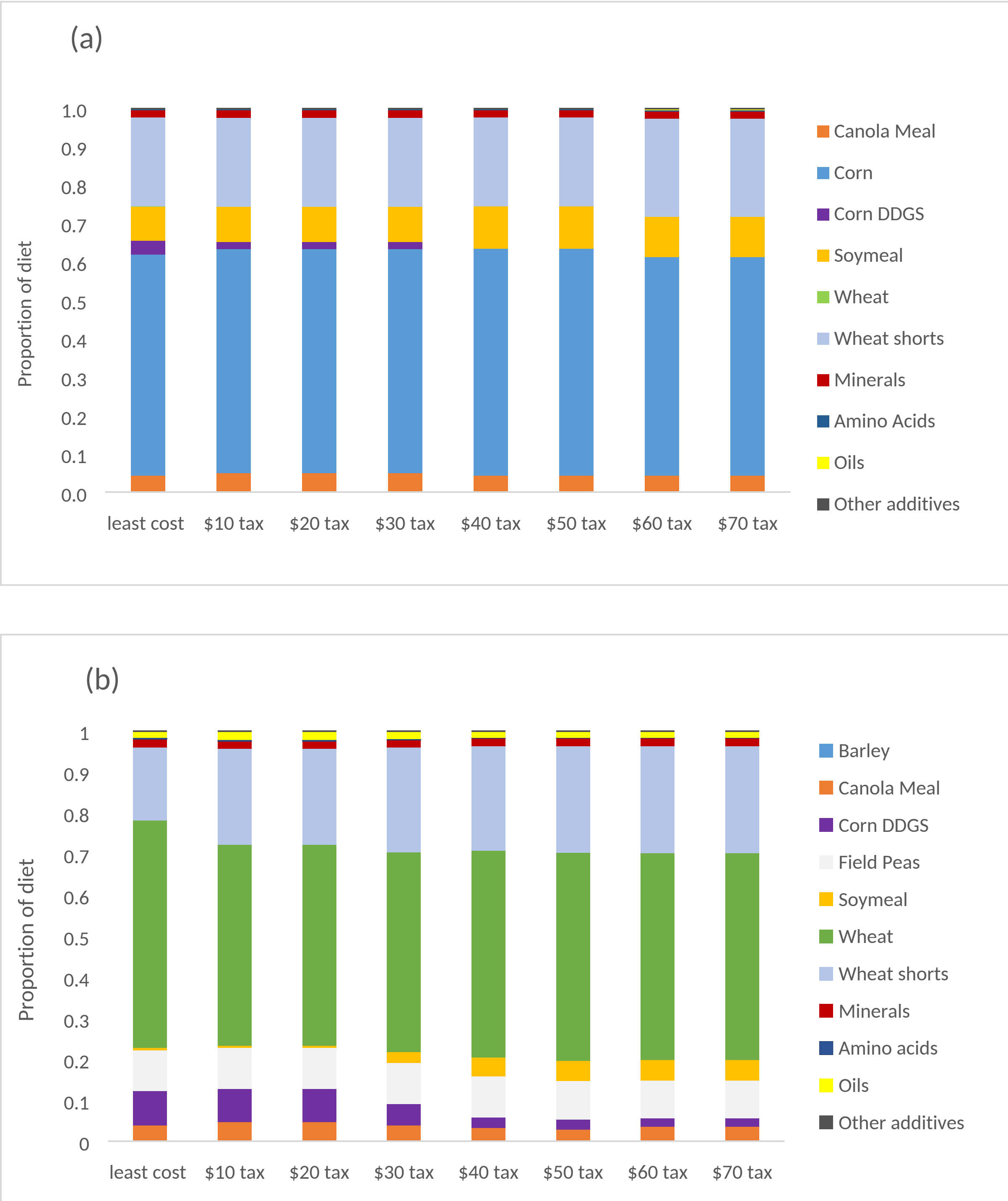
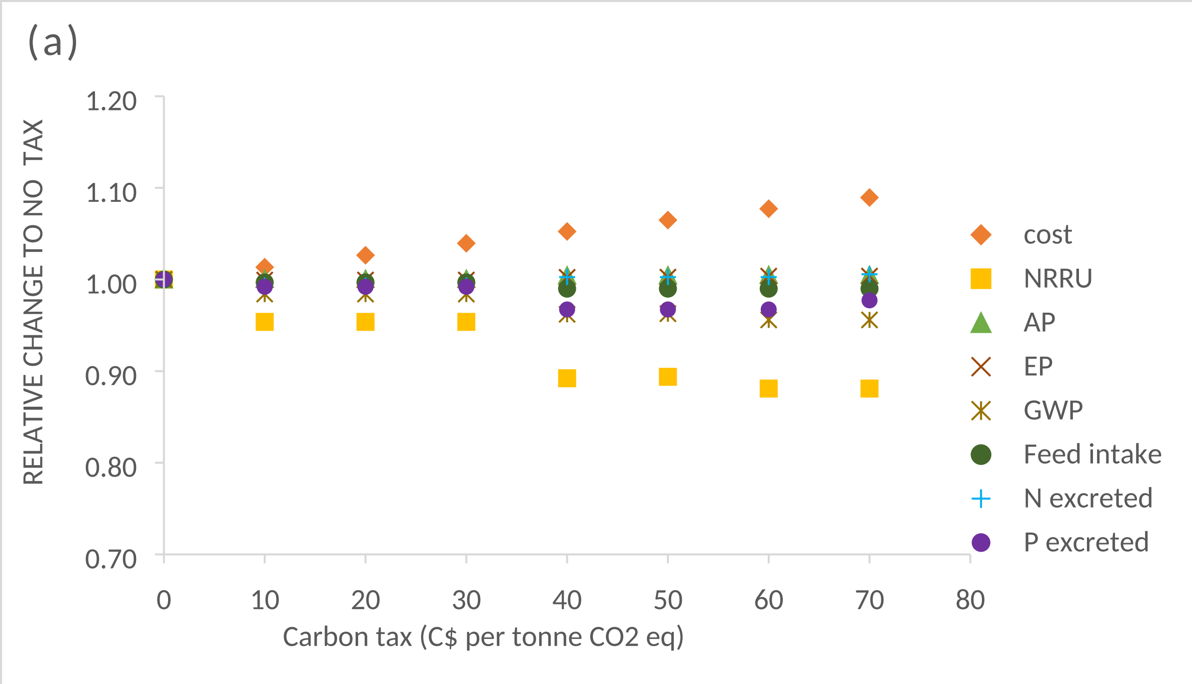


Figure 3 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent. C$ = Canadian Dollars.



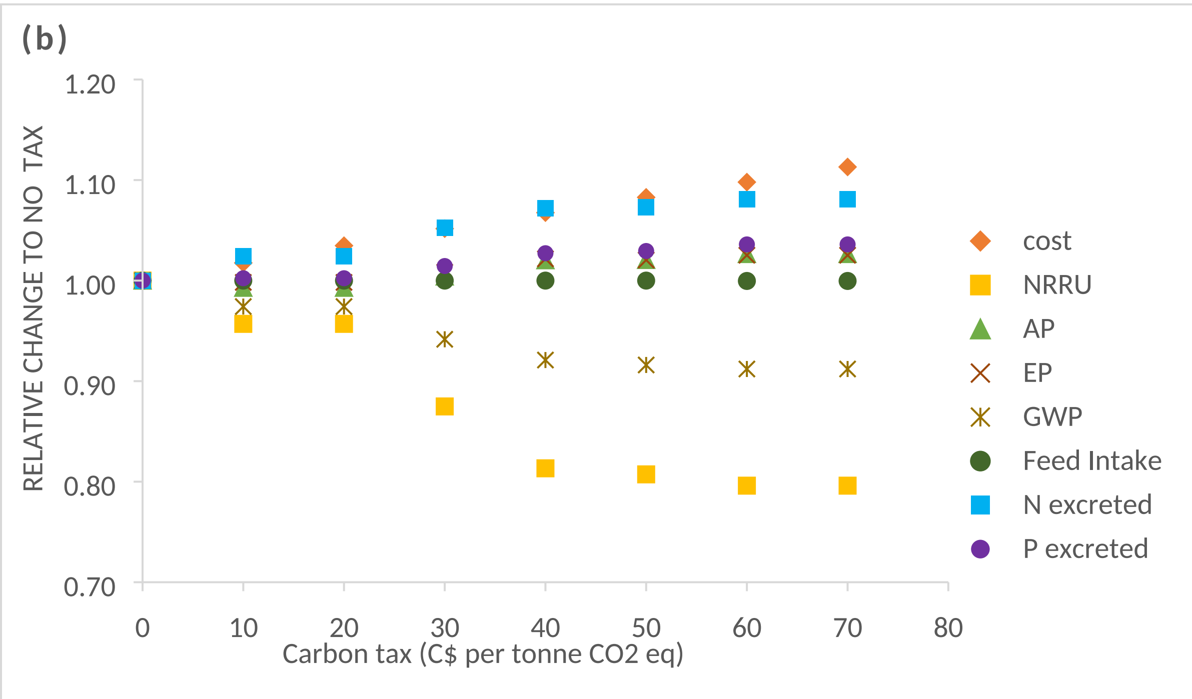


Figure 4 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least-cost subject to different levels of carbon tax in a) Eastern Canada; b) Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq). NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential. C$ = Canadian Dollars.

Figure 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$40 in Western Canada. Despite this, the general trend in both scenarios as tax levels increased was a diminishing reduction in GWP caused by the carbon tax per C$ cost increase. 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.

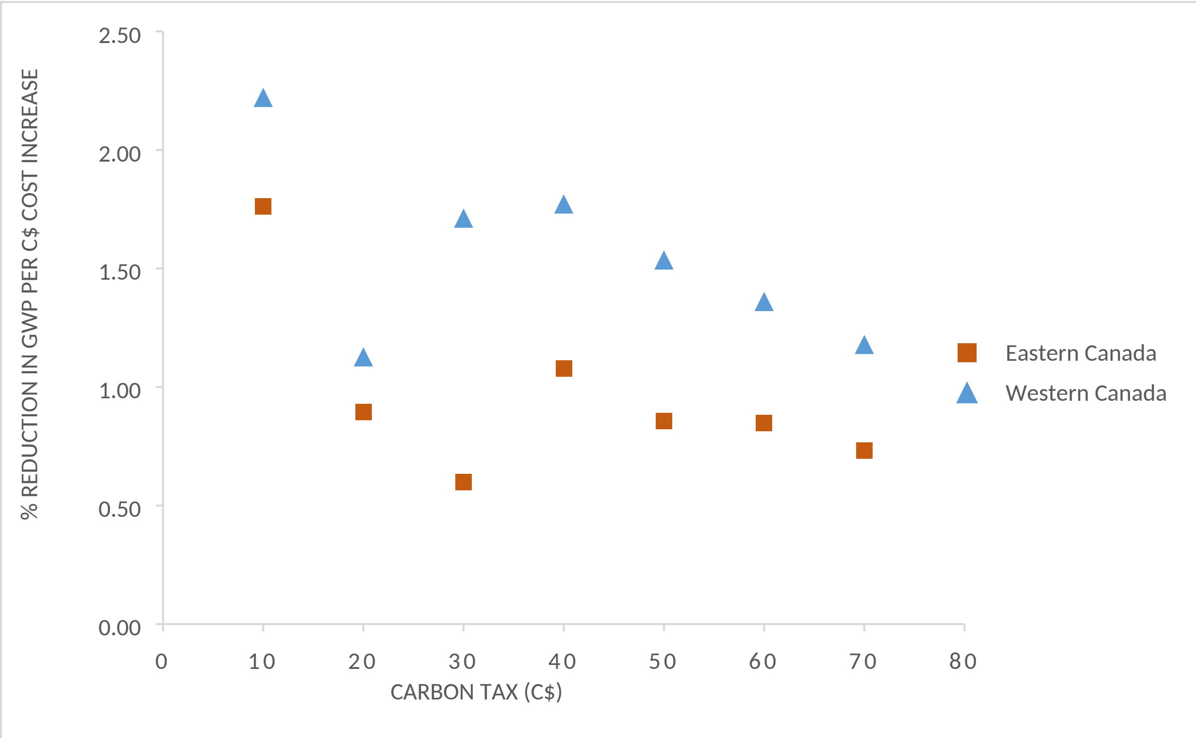


Figure 5 The relative reduction in GWP per C$ increase in feed cost caused by pig diets formulated for least-cost and subjected to different levels of carbon tax compared to no-tax scenarios modelled for Eastern Canada and Western Canada. Carbon tax levels are shown in C$ per tonne of CO_2 equivalent (eq.) C$ = Canadian Dollars.

3.2 Nitrogen tax

The ingredient compositions of the diets formulated at different levels of taxation are shown in figures 6a and 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 are shown in figures 6a and 6b 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. As tax levels increased they added to feed costs incrementally 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. As N tax increased, the inclusion of ingredients with relatively low levels of environmental impact per kg (see Table 1), such as wheat shorts and soymeal, were reduced. Levels of corn, wheat and synthetic amino acids in the grower/finisher diets (which were associated with higher environmental impact levels per kg) all increased in order to reduce the crude protein level and amino acid content of the diet and reduce 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.

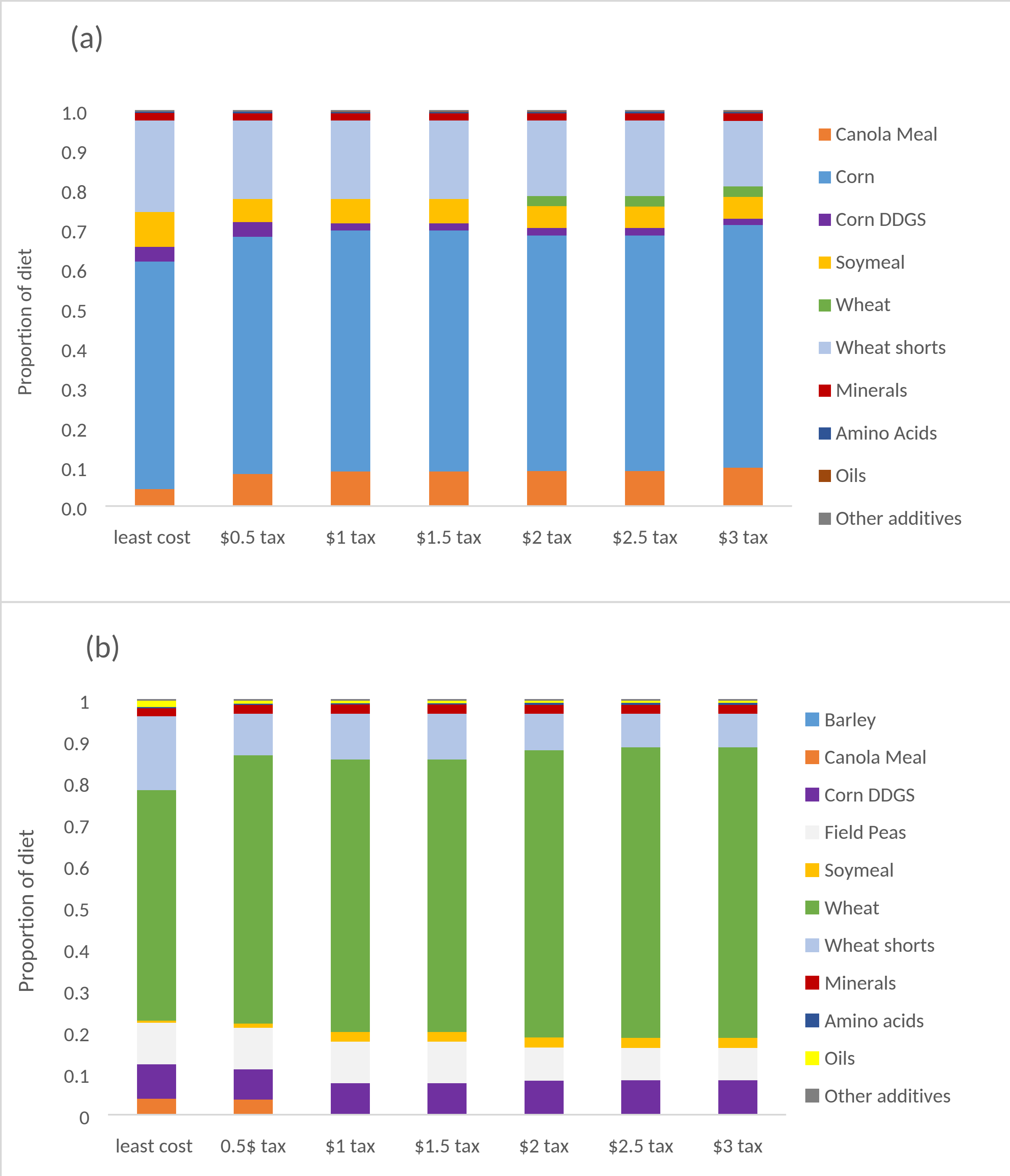


Figure 6 The overall ingredient and nutritional composition (across all 4 feeding phases) of grower/finisher diets formulated at different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field in manure. C$ = Canadian Dollars.

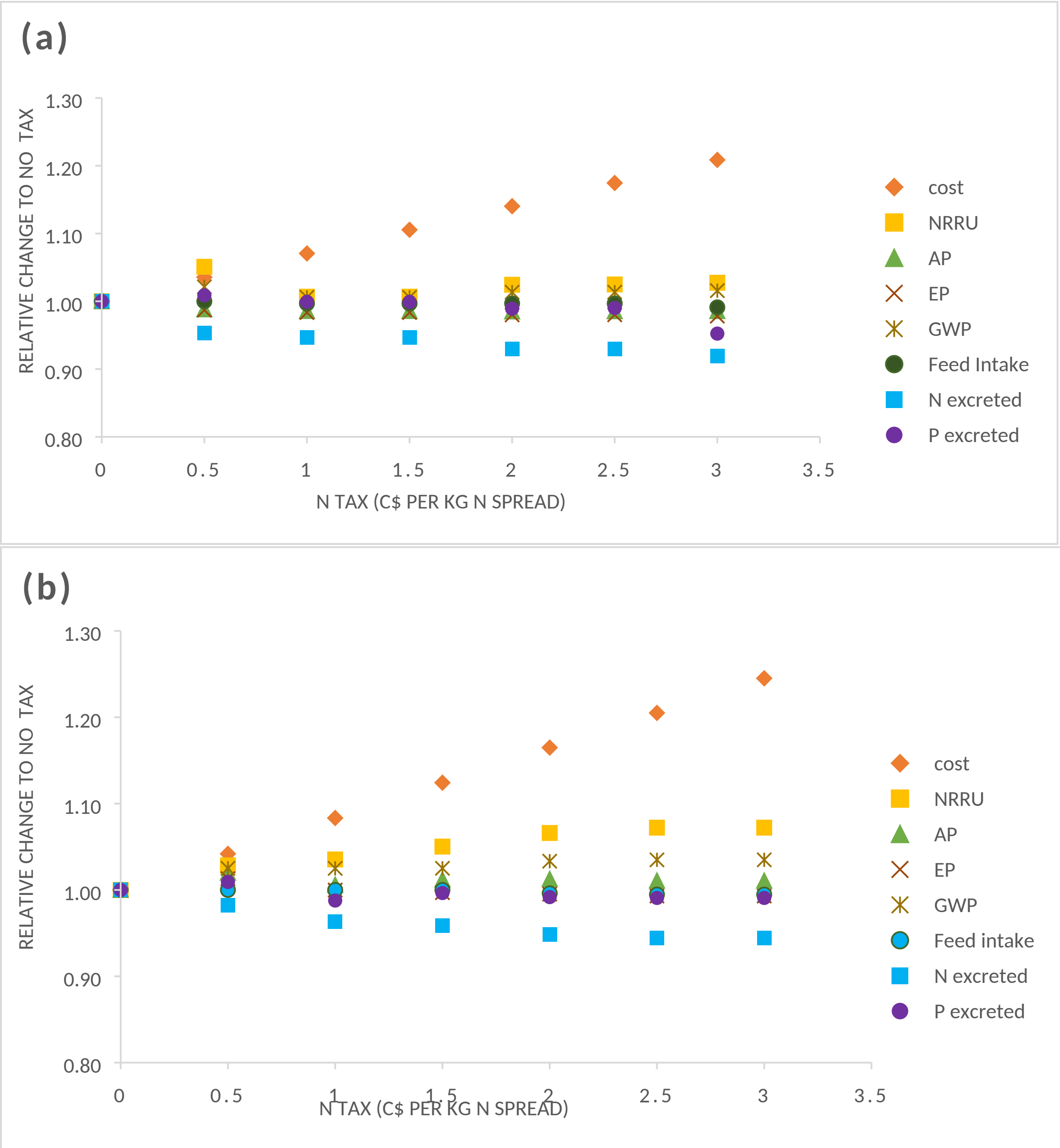


Figure 7 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least-cost subject to different levels of nitrogen tax in a) Eastern Canada; b) Western Canada. Nitrogen tax levels are shown in C$ per kg of N spread to field 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, at 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, soymeal, synthetic amino acids and animal-vegetable oil all increased, 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 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 diminished, i.e. as N tax increased it became less cost effective to reduce 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 7a &7b), and the N tax was able to make greater reductions in N excretion for a lower relative increase in cost in the East Canadian scenario.

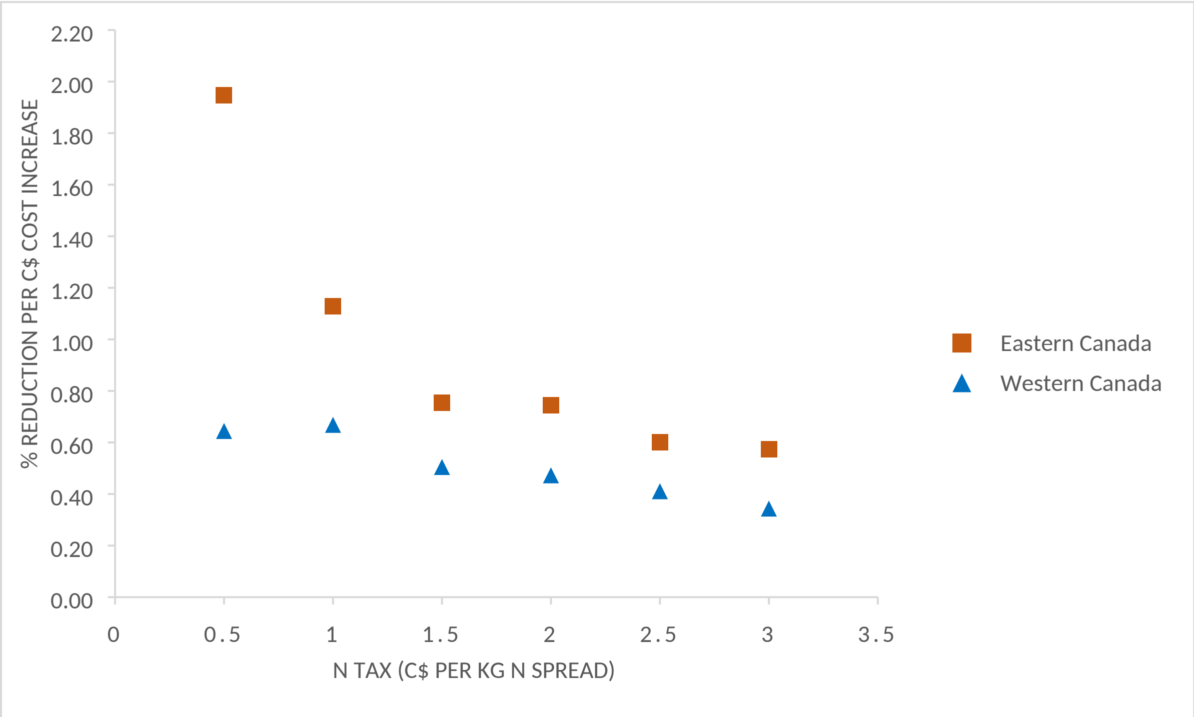


Figure 8 The relative reduction in N excretion per C$ increase in feed cost caused by pig diets formulated for least-cost and subjected to different levels of N tax compared to no-tax for scenarios in Eastern Canada and Western Canada. N tax levels are shown in C$ per kg of N in manure spread to field. C$ = Canadian Dollars.

3.3 Phosphorus tax

The ingredient compositions of the diets formulated at different levels of P taxation are shown in figures 9a and 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 10a and 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 significantly 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 increases of 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 model 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 1) 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 soymeal (both higher in AP and EP than wheat shorts per kg, Table 1) 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 the 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 grower/finisher 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 dried distillers grains with solubles 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 1) caused the reduction in AP, and increased corn dried distillers grains with solubles inclusion increased NRRU.

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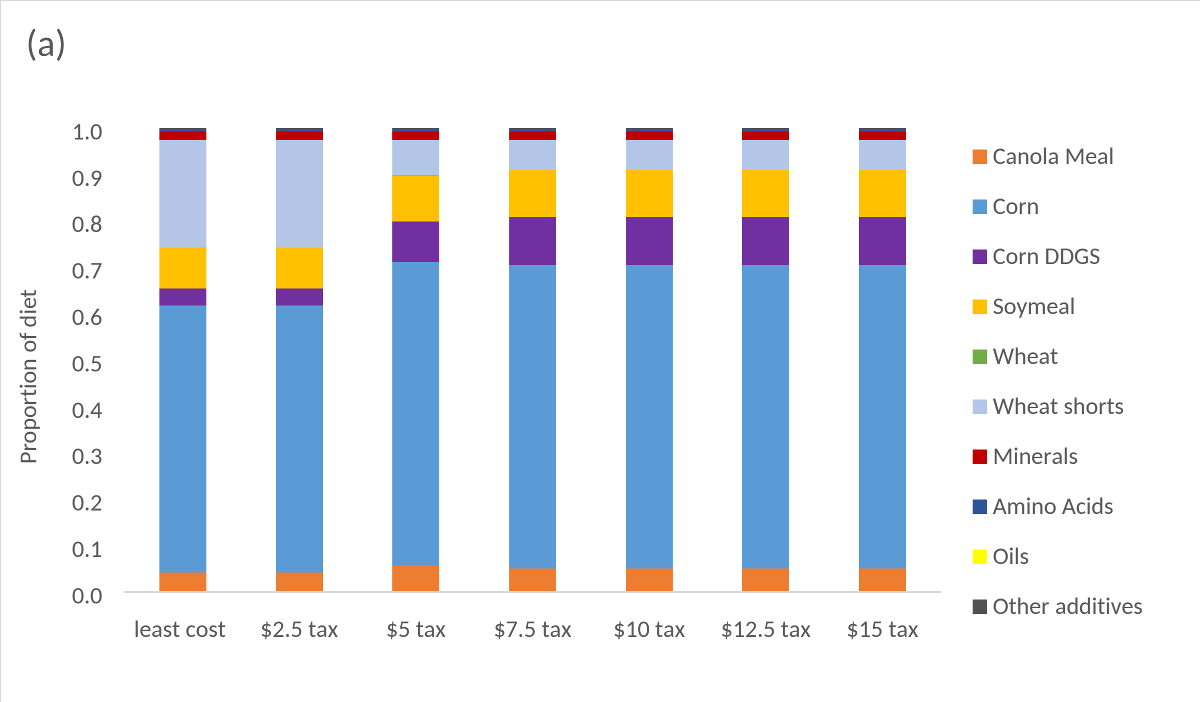

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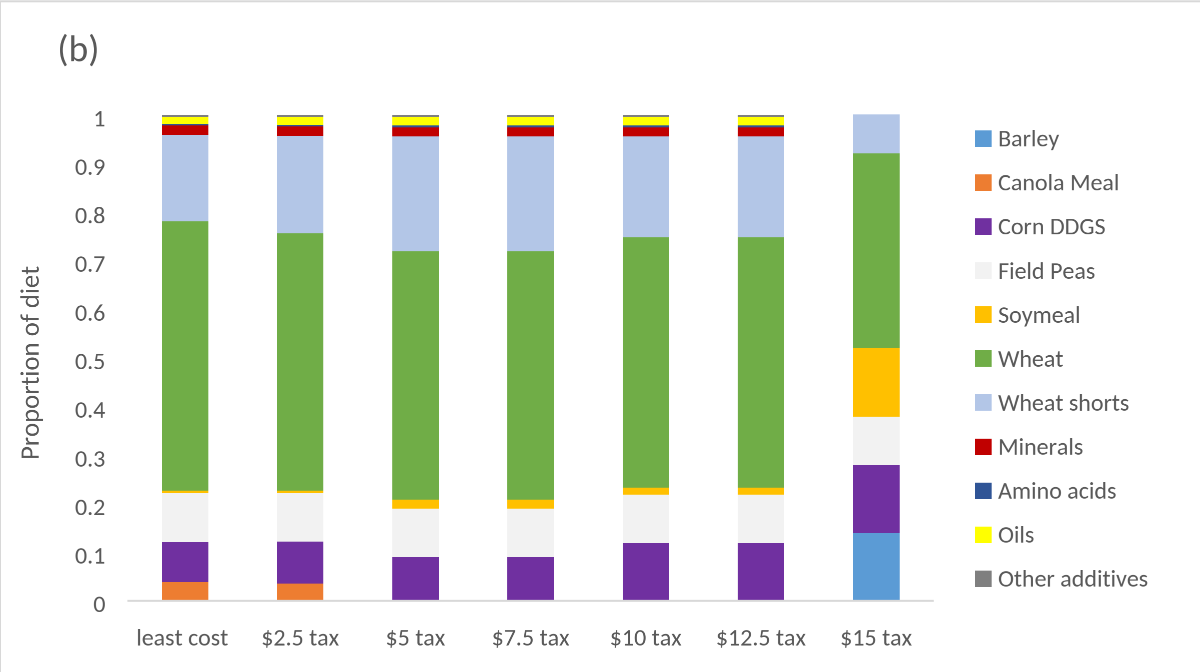

480 **Figure 9** The overall ingredient and nutritional composition (across all 4 feeding phases) of
481 grower/finisher diets formulated at different levels of phosphorus tax in a) Eastern Canada; b)
482 Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in
483 manure.

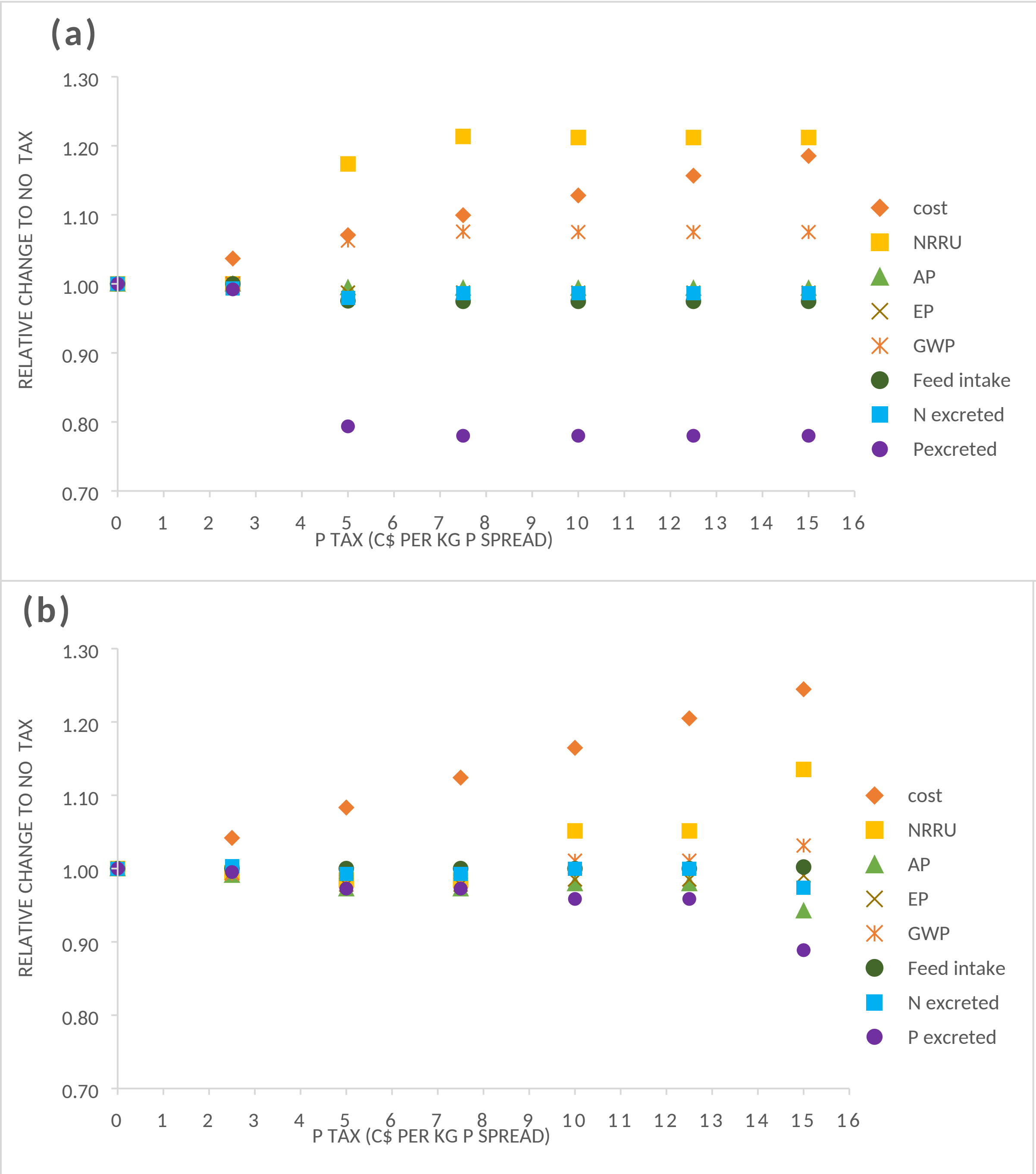

Figure 10 The relative levels of feed cost, feed intake, nutrient excretion and environmental impacts resulting from pig diets formulated for least-cost subject to different levels of phosphorus tax in a) Eastern Canada; b) Western Canada. Phosphorus tax levels are shown in C$ per kg of P spread to field in manure NRRU = Non-renewable resource use, AP = Acidification Potential EP = Eutrophication Potential, GWP = Global Warming Potential.

Figure 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 10a & 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.

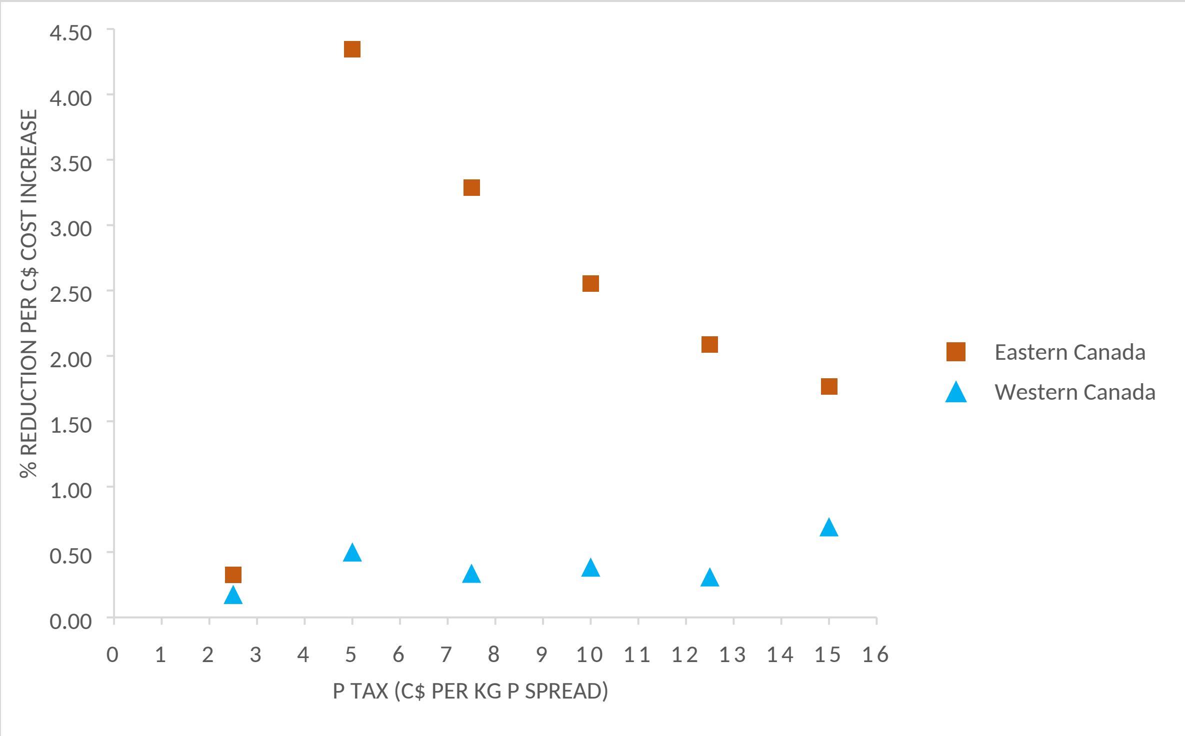

Figure 11 The relative reduction in P excretion per C$ increase in feed cost caused by pig diets formulated for least-cost and subjected to different levels of P tax compared to no-tax for scenarios in Eastern Canada and Western Canada. P tax levels are shown in C$ per kg of P in manure spread to field. C$ = Canadian Dollars.

4. 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 environmental impacts (Steinfeld et al., 2006). For example, recent focus on the contribution of livestock production to GWP has led to a Danish think tank recommending a carbon tax on livestock products in order to alter eating habits (Withnall, 2016). The production of feed materials and the storage and disposal of manure are generally the most important considerations regarding the environmental impacts of non-ruminant livestock production systems (Basset-Mens and Van Der Werf, 2005; Leinonen et al., 2012; Williams et al., 2006). 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 presented in this study demonstrate that when nutrient excretion is reduced through the introduction of specific taxes, it does not necessarily follow that overall levels of environmental impact have been reduced in pig farming systems. In this study we developed a novel framework to investigate the potential effect of 3 different environmental taxes on diet-formulation in pig farming systems, and the implications for the environmental impacts of the production system using multiple impact categories. While previous studies have integrated taxes on P excretion in pig systems (Pomar et al., 2007) and methane emissions in dairy systems (Moraes et al., 2012) into diet-formulation exercises, neither considered the implications for multiple environmental impact categories at the system-level.

Although we used Canadian pig farming as an example system, our findings have broader implications for decision making including diet-formulation across the agriculture sector, in any country/region where environmental taxes such as those tested here are implemented.

Expanding the approach to a larger study considering feed decision across livestock systems for different species may provide a more holistic assessment of the effects of such policies. Nevertheless, the more focused analysis reported here has demonstrated some of the real challenges facing policy makers seeking to reduce the environmental externalities of food production systems. The environmental taxes were tested in two regional scenarios which differed in ingredient availability and prices, as well as typical manure management practices. Environmental taxes levied at the same rates had different implications for environmental impact in the two regional scenarios. For example the carbon tax was able to produce greater reductions in GWP in the scenario for Western Canada than in the East, and did so more efficiently in terms of relative reduction in GWP per C$ cost increase. The results showed the importance of considering spatial differences when assessing the potential implications of environmental taxes. A policy which may be able to greatly reduce environmental impacts in one region may have very little potential to do so in another while still increasing costs for the affected farmers.

It was hypothesised that in each case as the environmental taxes increased, they would reduce levels of their targeted emission type by altering the least-cost formulation. As described in the results above, this was the case for each tax, i.e. GWP caused by feed production was reduced by the carbon tax, N excretion was reduced by the N tax and P excretion was reduced by the P tax. This agrees with the findings of previous diet-formulation exercises which have integrated levies on P excretion in pig systems (Pomar et al., 2007), and a carbon tax on methane emissions in dairy systems (Moraes et al., 2012). 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 was only one or two such trigger points at the lower end of the tax 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. Analyses 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 optimising energy markets and diet-formulation is that there is 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 for such policies.

In the case of policies such as a carbon tax, the tax assigned to commodities is based on a calculation of the GWP caused by its production. In LCA models of animal feed supply chains, calculating the environmental impact of different feed ingredients often requires co-product allocation as, for example, crop systems often produce multiple products which generate revenue. Co-product allocation based on the economic value of each co-product is currently advised by the FAO for animal feed supply chains (FAO, 2014). However, there is the theoretical possibility that environmental policies such as a carbon tax, may have an indirect effect on ingredient prices beyond the direct additional cost of the tax. This could 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. For economic allocation in LCA models, the issue of price variability can be greatly reduced by using multiyear averages of commodity prices (Guinée et al., 2004). However, for economic allocation the problem of such taxes indirectly affecting prices, such as increasing the market price of co-products with low carbon footprint is 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.

As shown in the results, individual policy measures are likely to have spill-over effects, perhaps causing increases in other types of environmental impact. For all of the taxes evaluated (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 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 grower/finisher 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 (Mackenzie et al., 2016a). 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 discrepancy between the results of these two approaches also allude to more general issues in the design of environmental taxes. Environmental taxes are most easily levied on aspects of a production system which are easily measured such as the amount of N or P spread to land in manure or (less easily) the carbon footprint of materials purchased. However, such simple approaches cannot capture complex interactions in these systems where, for example, targeting reductions in one area of the system such as reducing the carbon footprint of feed may cause increased impacts elsewhere in the system in this case from nutrient excretion. This makes it difficult for policy makers to design taxes in a way which can cause large reductions in the environmental impact of agricultural systems.

Levying simple flat taxes on aspects of the system, such as nutrients spread to land, also assumes that all manure applied to land has equal potential to cause negative environmental impacts. This is of course not the case, as factors such as soil type, time of year, application method and the concentration of agricultural activity are all important in determining the potential benefits and harms of applying manure to land. However, accounting for each of these factors adds layers of administrative complexity to any potential environmental tax. A pragmatic policy response might comprise a tiered (rather than flat rate) taxation system to accommodate the non-linear nature of environmental externalities. Emission intensity thresholds would need to be established and progressively higher marginal tax rates applied for each successive threshold increase in emissions. This system would have the advantage of progressively penalising sub-optimal activities and technologies that are responsible for the largest negative externalities. Businesses that have lower environmental footprint would incur a reduced marginal tax rate or may be exempt entirely if their emissions intensity is below a designated threshold. The perceived fairness of any environmental tax reflects normative judgements of policy makers and their electorates. The aforementioned system of progressive tax rates may be deemed more acceptable as it places greatest burden on the heaviest 'polluters'. However, environmental taxes could threaten the viability of some businesses; especially if those businesses are exposed to competition from imported products that might not have incurred similar environmental taxes. Moreover, farm-level impacts will vary according to business characteristics (resources, technology, management, etc.) as well as location (in terms of available feeds, relative prices). It would be very difficult to design systems of environmental taxes that are responsive to such heterogeneities, although policy makers could ring-fence the tax revenue for supportive initiatives within the affected industry or region. This might include measures to aid business adjustment and support technological innovation to reduce environmental impacts.

The results of this study 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 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.

5. 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 analysed. However, all taxes were effective in reducing the specific emission which was directly taxed; system-level environmental impact modelling can give perspective on whether potential environmental taxes are capable of reducing environmental impacts in livestock systems. The results also demonstrated that reduced nutrient excretion through dietary change does not necessarily reduce environmental impacts in livestock systems.

Finally, 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 in a scenario which only considered feed for pigs. 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.

Acknowledgements

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How effective can environmental taxes be in reducing the environmental impact of pig farming systems? _ Abstract only

Abstract: Environmental taxes are a form of incentive regulation available to governments to drive reductions in environmental impact. The aims of this study were to: 1) develop a framework that enabled quantification of the potential effect of environmental taxes on pig diet composition and 2) examine the relationship between tax level and its effectiveness in reducing environmental impacts from pig systems. Three taxes were investigated: a carbon tax on the feed ingredients as purchased, and two financial penalties on the field spreading of N and P in manure respectively. Each tax was integrated into a diet-formulation model 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 spatial variation in the consequences of tax measures on diet-formulation. 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, acidification potential, eutrophication potential and non-renewable resource use. As environmental tax levels increased, trigger points in the tax range caused dietary change which reduced levels of the targeted emission type. In almost all the tax scenarios the largest reductions in the target emission per Canadian dollar (C$) increase in cost were achieved at the lower end of the tax range tested, as diminishing marginal returns were evident. The taxes on spreading N and P in manure did not significantly reduce levels of any environmental impact category tested in most 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 considered. These results showed the potential for taxes which target specific emissions, to increase system-level environmental impacts in livestock production. The study demonstrated how system-level environmental impact models can be used to quantify the potential of environmental taxes set at different rates to reduce overall environmental impact levels in livestock systems.

How effective can environmental taxes be in reducing the environmental impact of pig farming systems? – Highlights

- The ability of taxes to reduce the environmental impact pig systems was investigated
- The environmental impacts were assessed through a Life Cycle Assessment model
- Taxes on carbon footprint, N and P excretion were tested at incremental tax levels
- Reductions in C footprint via the C tax caused increases in other impact categories
- N & P taxes did not reduce any of the environmental impact categories tested

How effective can environmental taxes be in reducing the environmental impact of pig farming systems? – Supplementary Material

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Appendix 1: Nutritional rules used to formulate diets

A linear programming algorithm for diet-formulation was used to formulate grower/finisher diets for each taxation scenario; the diet-formulation rules used are described in detail in Mackenzie et al. (2015). In this study all diets were formulated to minimise feed cost per kg live weight gain (least-cost) for the grower/finisher phase in each tax scenario. The predicted start weight of the pigs in the diet-formulation model was fixed at 27.4 kg with a finish weight of 124 kg for the grower/finisher phase, based on benchmark data collected for a previous LCA study of Canadian pig farming (Mackenzie et al., 2015). Minimum nutrient levels in g/MJ of Net Energy were defined for each feeding phase, so that the protein and macronutrient content of the feed would not be limiting for animal growth (NRC, 2012); it was thus expected that feed intake enabled animals to meet their energy requirements (Kyriazakis and Emmans, 1995; Patience, 2012). Thus the average predicted Net Energy intake was constant for all diets. The main nutritional specifications of the “typical” Canadian diet are found in Table A1 and it was assumed that this diet ensured an average gain: feed ratio of 0.365 kg/kg based on data collected for a previous LCA study of Canadian pig farming (Mackenzie et al., 2015). 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 grower/finisher diet for each phase can also be found in Table 1.

Table A1 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

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 (Mackenzie et al., 2016b). These were based on advice on diet-formulation for pigs from the Ontario Ministry of Agriculture Food and Rural Affairs (OMAFRA, 2012), the full list can be found in Table A2. Nutritional values for all ingredients in the diets were primarily taken from the Stein Monogastric Nutrition Laboratory ingredient matrix (Stein Monogastric Nutrition Laboratory, 2014).

Table A2 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, 2012) as well as peer reviewed studies in the case of some important co-products(Mackenzie et al., 2016).

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
Soya 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
Soy 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

Appendix 2: Regional price ratios used for diet formulation

Table A3 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, 2014)).

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
Soya 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
Soya 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 3: Sources for Life Cycle Inventory data ingredients

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). 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). 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 2015).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. The sources of data for other minor ingredients included in table A3

Table A4 Minor ingredients LCI data sources

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

Table A5 shows a summary of how co-product allocation was carried out for multi-output processes in the feed supply chain

Table A5 Allocation factors used for multi-output processes in the feed supply chain

Multi-output 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 4: On farm energy use data

Table A6 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 5: 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 A8.

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 A1 (Intergovernmental Panel on Climate Change 2006). CH4 emissions from manure during housing were considered to be negligible.

*Equation A1: 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 A2 (Intergovernmental Panel on Climate Change 2006).

Equation A2: $EF = VS * B_0 * 0.67 kg/m^3 * MCF_{S,k} * MS_{S,k}$

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 A3 (Intergovernmental Panel on Climate Change 2006)

Equation A3: $VS = (GE * (1-DE) * (UE*GE)*(1-ASH/18.45))$

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 A7 assumptions regarding storage type (Statistics-Canada 2003; Sheppard, Bittman, Swift, et al. 2010)

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 A4

Equation A4: 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 A5

*Equation A5: 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 A7 taken from (Sheppard, Bittman, Swift, et al. 2010) – 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, Bittman, Swift, et al. 2010). TAN mean value was 70% N excreted with a range of 0.62-0.79

Table A8 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 A6: 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 A7: $EF\_NH3\_Sl = 0.13*(1-0.058*(15-T))$

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 A8: $EF\_NH3\_So = 0.13*(1-0.058*(17-T))$

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, Bittman, Swift, et al. 2010)

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 A9.

Equation A9: $N\_Loss\_App = N\_loss\_app\_M - (0.75*N\_loss\_app\_s)$

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 A10: $N\_Loss\_app\_M = (Napp*EFm\_NH3\_app\_) + (Napp*EFm\_N2O\_app) + (Napp*EFm\_NOx\_app)) + (Napp*EFm\_NO3\_app)$

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 A11: $N\_Loss\_app\_s = (Napp*EFs\_NH3\_app) + (Napp*EFs\_N2O\_app) + (Napp*EFs\_NOx\_app) + (Napp*EFs\_NO3\_app)$

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 A8 for ranges of all parameters).

The increased emissions of NH3 and N2O account for most (~19% - see emission factors table A8) 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 A8).

Emission Factors

Table A9 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, Bittman, Swift, et al. 2010)
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	(Rochette et al. 2008; Bouwman et al. 2002)
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, Bittman, Swift, et al. 2010)
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	(Rochette et al. 2008; Intergovernmental Panel on Climate Change 2006)
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	(Rochette et al. 2008; Bouwman et al. 2002)
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, Bittman & Bruulsema 2010)

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	(Rochette et al. 2008; Intergovernmental Panel on Climate Change 2006)
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	0.2	0.1	0.4	0.2	0.1	0.4	(Prapaspongsa et al.

	N stored in unlined lagoon							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, Bittman, Swift, et al. 2010)
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 6: Further description of the uncertainty analysis

The uncertainty in the environmental impact calculations was quantified using Monte-Carlo simulations. 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, whereas 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 2 systems, used to calculate feed intake, would be considered α uncertainties. Variability in all characteristics of herd performance other than feed intake was assumed to be independent of feed composition in the grower/finisher 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. Major causes of uncertainty in the LCA model of pig farming systems included; the feed: gain ratio achieved by animals for a specific diet, some emission factors used in the manure model, and an aspect of the manure model which estimated the effectiveness of the organic manure in replacing the need to use inorganic fertilizer in future crop production (Mackenzie et al. 2015).