Appraisal of the Environmental Sustainability of Milk Production Systems in New Zealand

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Jeerasak Chobtang

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Executive Summary

Life Cycle Assessment (LCA) plays an important role in the environmental assessment of agricultural product systems, including dairy farming systems. Generally, an LCA study accounts for the comprehensive resource use and environmental emissions associated with the life cycle of a studied product system. The inventoried inputs and outputs are then transformed into different environmental impact categories using science-based environmental cause-effect mechanisms. There are different LCA modelling approaches (e.g. attributional LCA [ALCA] and consequential LCA [CLCA]) that can be used to address different research questions; however, there is currently no consensus on the most appropriate approach and when to use it. These LCA approaches require different types of data and methodological procedures and, therefore, generate different sets of environmental information which may have different implications for decision-making.

In the present research, a series of studies utilising different LCA modelling approaches were undertaken of pasture-based dairy farming systems in the Waikato region (the largest dairy region in New Zealand). The purposes of the studies were to: (i) assess the environmental impacts and identify environmental hotspots of current pasture-based dairy farming systems, (ii) compare environmental hotspots between high and low levels of dairy farm intensification, (iii) investigate the environmental impacts of potential alternative farm intensification methods to increase milk productivity, and (iv) assess the environmental impacts of different future intensified dairy farming scenarios. Twelve midpoint impact categories were assessed: Climate Change (CC), Ozone Depletion Potential (ODP), Human Health Toxicity - non-cancer effects (Non-cancer), Human Health Toxicity - cancer effects (Cancer), Particulate Matter (PM), Ionizing Radiation - human health effects (IR), Photochemical Ozone Formation Potential (POFP), Acidification Potential (AP), Terrestrial Eutrophication Potential (TEP), Freshwater Eutrophication Potential (FEP), Marine Eutrophication Potential (MEP) and Ecotoxicity for Aquatic Freshwater (Ecotox).

Firstly, the environmental impacts of 53 existing pasture-based dairy farm systems in the Waikato region were assessed using ALCA. The results showed that both the off-farm and on-farm stages made significant contributions to a range of environmental impacts per kg of fat- and protein-corrected milk (FPCM), and the relative contributions

of the stages varied across different impact categories. Farms classified as high intensification based on a high level of farm inputs (i.e. stocking rate, level of nitrogen (N) fertiliser and level of brought-in feeds) had higher impact results than low intensification farms for 10 of 12 impact categories. This was driven mainly by the off-farm stage, including production of brought-in feeds, manufacturing of agrichemicals (e.g. fertilisers and pesticides), and transport of off-farm inputs for use on a dairy farm. The exceptions were the environmental indicators PM, POFP, AP and TEP; their results were determined mainly by ammonia emissions from the on-farm activities.

Secondly, environmental consequences resulting from meeting a future increase in demand for milk production (i.e. 20% more milk production per hectare relative to that in 2010/11) by using different farm intensification scenarios for dairy farming systems in the Waikato region were assessed using CLCA. In this study, only technologies/flows that were actually affected by use of different intensification options to increase milk production were accounted for. The identified intensification methods were: (i) increased pasture utilisation efficiency, (ii) increased use of N fertiliser to boost on-farm pasture production, and (iii) increased use of brought-in feed (i.e. maize silage). The results showed that improved pasture utilisation efficiency was the most effective intensification option since it resulted in lower environmental impacts than the other two intensification options. The environmental performance between the other two intensification options varied, depending on impact categories (environmental trade-offs).

Thirdly, prospective ALCA was used to assess the environmental impacts of six prospective (future) dairy farming intensification scenarios in the Waikato region, primarily involving increased stocking rate, that were modelled to increase milk production per hectare by 50% in 2025. In this study, prospective (future) average flows that were derived from extrapolation were accounted for. The potential intensification scenarios were: (i) increased animal productivity (increased milk production per cow), (ii) increased use of mixed brought-in feed, (iii) improved pasture utilisation efficiency, (iv) increased use of N fertiliser to boost on-farm pasture production, (v) increased use of brought-in maize silage, and (vi) replacement of total mixed brought-in feed in the second scenario by wheat grain. The results showed that, apart from improved animal productivity which was considered the best option, improved pasture utilisation efficiency was the second environmentally-preferential option compared with other intensification options for pasture-based dairy farming systems in the Waikato region. There were environmental trade-offs between other intensification options.

The present research demonstrated that pasture-based dairy farming systems in the Waikato region contribute to a range of environmental impacts. More intensive farming systems not only have increased milk productivity (milk production per hectare) but also increased environmental impacts (per kg FPCM) in most environmental impact categories. Farm intensification options associated with improved farm efficiency (e.g. animal productivity or pasture utilisation efficiency) are promising as they have lower environmental indicator results (per kg FPCM) compared with other intensification methods. Increased use of off-farm inputs (e.g. N fertilisers and brought-in feeds) increases some, and decreases other, environmental indicator results. Therefore, decision-making associated with choice of alternative farm intensification options beyond farm efficiency improvements will require prioritisation between different environmental impacts and/or focusing on the ability of key decision-makers to effect change (for example, by distinguishing between local and global activities contributing to environmental impacts).

The present research has shown that different LCA modelling approaches can be used in a sequential manner to maximise the usefulness of environmental assessment. Initially, ALCA (based on current average flows) can be used to identify environmental hotspots in the life cycle of dairy farming systems. This will generate environmental information that can assist in selection of improvement options. Subsequently, the improvement options selected should be evaluated using CLCA (based on marginal flows). This will produce comparative environmental information resulting from implementing the selected improvement options, strategies or policies in relation to a non-implementation scenario, when the wider contribution of co-products is accounted for. Finally, prospective ALCA (based on future average flows) can be used to assess total or net environmental benefits.

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The success of this project is largely supported by Sarah McLaren. Sarah is a great supervisor. She has never given up in encouraging, motivating and guiding me in any ways to address any emerging problems over the course of the project. The contribution of a daily supervisor, Stewart Ledgard, is valuable. Stewart has spent all efforts to help me to complete the project. The guidance of Danny Donaghy, a co-supervisor, is appreciated. Danny's expertise is great, and has helped me to gain insight into pasture-based dairy farming systems.

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Abbreviations and Acronyms

ALCA	Attributional Life cycle Assessment
AP	Acidification Potential
Cancer	Human Health Toxicity (cancer effects)
CC	Climate Change
Cd	Cadmium
CF	Carbon Footprint
CFC-11	Trichlorofluoro-methane
CH_4	Methane
CLCA	Consequential Life Cycle Assessment
CO	Carbon monoxide
CO_2	Carbon dioxide
Cr	Chromium
CTUe	Comparative toxic unit for ecosystems
CTUh	Comparative toxic unit for humans
Cu	Copper
DLUC	Direct land use change
DM	Dry matter
ECM	Energy-corrected milk
Ecotox	Ecotoxicity for Aquatic Freshwater
FCE	Feed conversion efficiency
FEP	Freshwater Eutrophication Potential
FPCM	Fat- and protein-corrected milk
FU	Functional unit
GHG	Greenhouse Gas
GWP	Global Warming Potential
ILUC	Indirect land use change
H^+	Hydrogen ion
Ha	Hectare
Hg	Mercury
IR	Ionizing Radiation (human health effects)
K	Potassium
kBq	kilobecquerel

kg	kilogram
L	Litre
LCA	Life Cycle Assessment
LCI	Life cycle inventory
LCIA	Life Cycle Impact Assessment
LU	Land use
LUC	Land use change
MEP	Marine Eutrophication Potential
N	Nitrogen
NH_3	Ammonia
NH_4^+	Ammonium
Ni	Nickel
NMVOC	Non-methane volatile organic compounds
NO_2^-	Nitrite
NO_3^-	Nitrate
Non-cancer	Human Health Toxicity (non-cancer effects)
NO_x	Nitrogen oxides
ODP	Ozone Depletion Potential
P	Phosphorus
Pb	Lead
PKE	Palm kernel expeller
PM	Particulate Matter and Respiratory Inorganics
POFP	Photochemical Ozone Formation Potential
SO	Sulphur monoxide
SO_2	Sulphur dioxide
SO_3	Sulphur trioxide
t	tonne
***t*-test**	Student's t-test
TEP	Terrestrial Eutrophication Potential
VOCs	Volatile organic compounds
WSI	Water scarcity index
Zn	Zinc

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Chapter 1 Introduction

1.1 The international and New Zealand dairy industries

Milk from dairy cows is one of the most important livestock commodities and is being produced and consumed at an increasing rate across the world (Miller and Auestad 2013). It has been anticipated that the global consumption of milk and dairy products will increase by approximately 30% in 2024 relative to 2014 (OECD/FAO 2015).

There are many different types of dairy farming systems around the world, ranging from small-scale (subsistence) to globalized commercial systems (Douphrate et al. 2013; Gall 2013). The key drivers determining these differences are generally associated with: (i) availability and accessibility of natural resources, (ii) climatic and geographic conditions, (iii) market opportunities, and (iv) cultural differences. However, in general, dairy systems have been increasingly expanding and intensifying in recent years (Miller and Auestad 2013), resulting in a steady increase in global milk production (see Figure 1.1a).

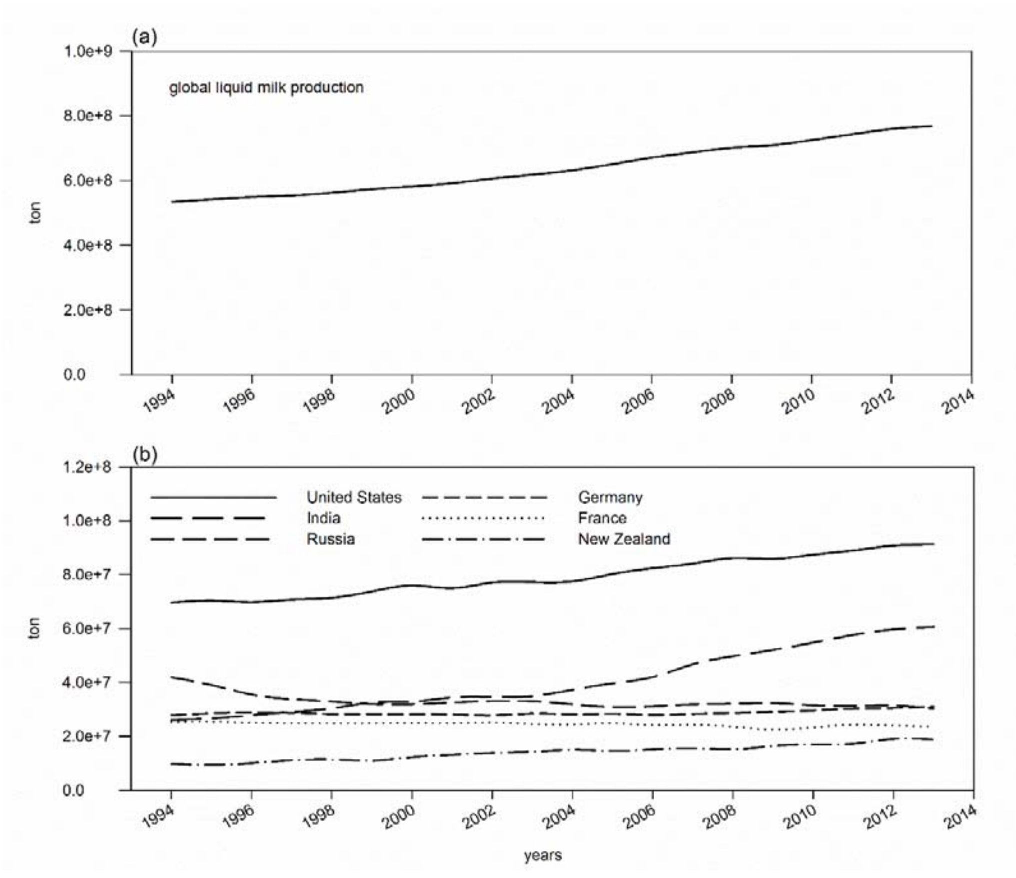

Figure 1.1 Trends in (a) global milk production, and (b) milk production in the top five leading countries (the USA, India, Russia, Germany and France) and in New Zealand over the past two decades (1994 - 2013). Source: FAO (2015).

It is noteworthy that the five leading milk production nations (the USA, India, Russia, Germany and France) show different trends in growth (see Figure 1.1b). Milk production in the USA and India has continuously increased, whereas milk production in Germany and France has remained unchanged over the past two decades. The latter situation is due mainly to the implementation of a milk quota system in Europe. For New Zealand, which is the 8th largest milk producing nation in the world (FAO 2015) and which does not operate a milk quota system, national milk production has steadily increased over the same period (see Figure 1.1b) (DairyNZ 2014).

Furthermore, milk and dairy products (e.g. cheese and whole milk powder) are a group of livestock commodities that are increasingly traded in international markets (Miller and Auestad 2013; Lagrange et al. 2015). The total quantities of globally exported cheese and whole milk powder (these two dairy products collectively represent the biggest share of internationally-traded dairy products (OECD/FAO 2013)), have substantially increased over the past two decades (see Figure 1.2).

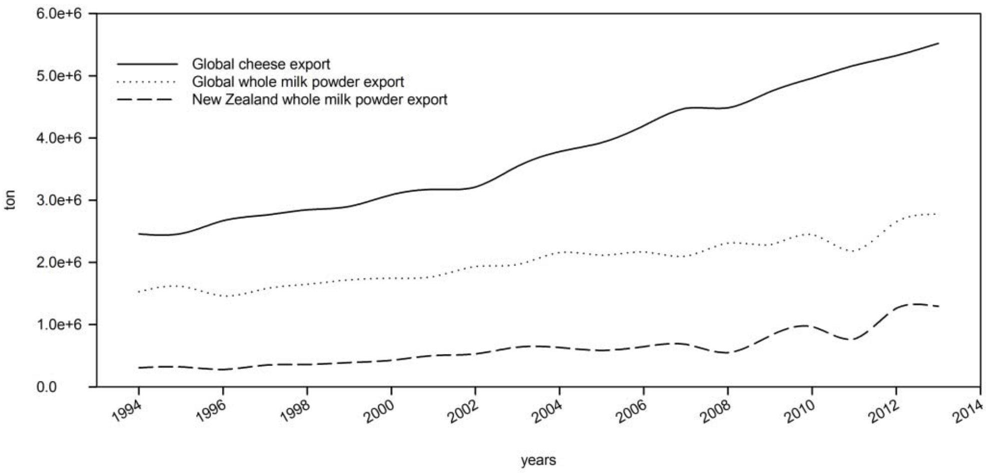

Figure 1.2 Trends in global exports of cheese and whole milk powder, and New Zealand export of whole milk powder over the past two decades (1994 - 2013). Source: FAO (2015).

Similarly, New Zealand‘s exports of dairy products over the same period have increased, especially whole milk powder which is the single largest exported dairy product of New Zealand (see Figure 1.2) (New Zealand Government 2013). Indeed, the New Zealand dairy sector generated 26% of total revenue from exports and accounted for 2.8% of national Gross Domestic Product in 2012 (New Zealand Government 2013).

The New Zealand Government has announced a national agenda aiming to double the monetary value of agricultural exports, including dairy products, by 2025 (New Zealand Government 2012). In addition to being supported by this national agenda, an expected increase in global demand for dairy products and their market prices is concurrently driving dairy farming systems in New Zealand toward more intensified systems where more off-farm inputs (e.g. fertilisers and brought-in feeds) are used to support increased farm productivity (milk production per hectare), and hence total milk production. However, it is currently unclear whether this change is leading to a net increase in environmental impacts of pasture-based milk production systems in New Zealand. For example, Tilman et al. (2011) reported that agricultural intensification generally results in a reduction in environmental impacts when calculated on a per-product basis whilst Nemecek et al. (2011) found that environmental impacts usually increase when computed on a per-hectare of farmland basis in Swiss agricultural systems. Thus the calculated environmental impacts associated with dairy intensification may vary depending on factors such as: (i) the activities included within the scope of a study, (ii) the unit of analysis for the study (e.g. product versus area farmed), (iii) the geographical scale of analysis, and (iv) the type and number of impact categories addressed (Basset-Mens et al. 2009; Guerci et al. 2013; Battini et al. 2016). Therefore, a robust assessment approach that can produce comprehensive environmental information and engage with different perspectives is needed.

1.2 Environmental scrutiny in New Zealand

In recent years, increased environmental scrutiny across the world has triggered consumers to favour environmentally-friendlier products (Echeverría et al. 2014; Zhao and Zhong 2015); this trend has particularly been observed amongst consumers in high-income countries (Fairbrother 2012). It has been reported that use of environmental sustainability labelling systems has a positive effect on consumer willingness to pay for livestock products (Echeverría et al. 2014; White and Brady 2014; Tait et al. 2016). For dairy products in particular, it has recently been reported that environmental labelling systems (e.g. carbon footprint) can lead to increased sales of dairy products in Sweden by 6% – 8% in 2013, compared with non-labelled products (Elofsson et al. 2016). In addition, Echeverría et al. (2014) reported that Chilean consumers are willing to pay 29% more on carbon-labelled (total GHG emissions) dairy products.

Even though the dairy sector contributes to improved global food security (Miller and Auestad 2013; Lagrange et al. 2015), it also contributes to adverse effects on the environment (i.e. exacerbated environmental degradation) (FAO 2006; Gerber et al. 2011). In New Zealand, dairy farming has been identified as a major contributor of greenhouse gas (GHG) emissions, accounting for 45% of total GHG emissions from the agricultural sector, equivalent to 32% of national GHG emissions in 2013 (Ministry for the Environment 2015), and of freshwater pollutants (Monaghan and Smith 2012; Scarsbrook and Melland 2015). In the Waikato region of New Zealand, for example, dairying is responsible for 68% and 42% of total nitrogen (N) and phosphorus (P) emissions respectively, entering natural water bodies (Environment Waikato 2008). Researchers have claimed that intensification of dairy farming systems in New Zealand using more off-farm inputs (e.g. fertilisers and feeds) causes an increase in a range of environmental impacts (MacLeod and Moller 2006; Foote et al. 2015).

In short, the New Zealand dairy sector plays an important role not only in the national economy but also in environmental degradation. Therefore, it is important to understand the comprehensive environmental impacts associated with intensification of dairy systems, and identify environmental hotspots[1] where mitigation options or improvement options can be applied, in order to reduce these environmental impacts.

1.3 Life Cycle Assessment

Life Cycle Assessment (LCA) is a systematically analytical approach for assessing environmental impacts associated with the life cycle of a product (International Organization for Standardization 2006a, 2006b). The main advantages of this approach are: (i) it accounts for environmental impacts along the entire supply chain of a studied product, and (ii) it provides multiple environmental indicator results. The former advantage means that adverse effects associated with burden-shifting between life cycle stages, and between relevant businesses, can be identified and (hopefully) avoided. The latter advantage facilitates decision makers to take into account all relevant environmental aspects when making decisions to more comprehensive environmentally-sustainable development (Finnveden et al. 2009; Hellweg and Milà i Canals 2014).

[1] An individual life cycle stage(s)/process(es) or contributing substance(s) in the life cycle of a product that makes the most significant contribution to one or more environmental impact category results.

For the dairy sector, the LCA approach is increasingly recognised as an important tool for assessing the life cycle environmental impacts of dairy products (FAO 2010). The International Dairy Federation (International Dairy Federation 2010) has developed a guideline for performing environmental assessment of dairy products. However, the guideline is focussed on just a single environmental impact, i.e. total GHG emissions or the carbon footprint. In the absence of guidelines for assessment of all relevant environmental aspects, there is potential for burden-shifting between environmental impacts, life cycle stages and relevant businesses.

However, the use and interpretation of LCA results is not straightforward, especially when used to support decision- or policy-making (Weidema et al. 2009). In 2010, the international organisation, the European Commission (EC) - Joint Research Centre (JRC) - Institute for Environment and Sustainability (IES), published the International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC-IES 2010) as an international guideline to assist in performing LCA studies. The guideline assists in defining the type of representative data required (e.g. average or marginal data) and the appropriate methods to handle co-products (e.g. allocation or system substitution) with regard to LCA studies. Two LCA modelling choices (attributional LCA modelling [ALCA] or consequential LCA modelling [CLCA]) have been developed. These two approaches address different research questions. However, there are a number of different interpretations of when to use these approaches, and this is now a topic of hot debate (Ekvall et al. 2016). Therefore, there is a need for research to examine the implications of the use of these different approaches in order to support development of a consensual methodology to enhance the credibility and usefulness of LCA studies.

1.4 Study goals, research questions and objectives

This research used LCA to assess the environmental sustainability of pasture-based dairy farming systems using different farm intensification methods in New Zealand.

The overall goals of the research were defined as: (i) to develop a better understanding of the environmental impacts associated with current dairy systems, and (ii) to evaluate future intensification methods for dairy systems with respect to their environmental impacts.

The research was undertaken using a case study of dairy farming systems in the Waikato region, the single largest dairy region in New Zealand. The following research questions were defined:

1) What are the environmental impacts of current pasture-based dairy farming systems in the Waikato region, and are there differences between low and high levels of farm intensification?
2) What are the environmental consequences of an increase in demand for pasture-based milk production through the use of alternative farm intensification methods in the Waikato region?
3) What are the potential environmental impacts of future pasture-based dairy farming systems in the Waikato region?

The main objectives of the present study were therefore:

1) To assess the environmental impacts and identify environmental hotspots for the life cycle of current pasture-based dairy farming systems in the Waikato region (Chapter 3).
2) To compare environmental impacts between two quantitative contrasting levels (low versus high) of farm intensification in existing pasture-based dairy farming systems in the Waikato region (Chapter 4).
3) To explore the environmental consequences and potential environmental trade-offs associated with increased pasture-based milk production through the use of different farm intensification methods in the Waikato region (Chapter 5).
4) To assess potential environmental impacts stemming from future pasture-based dairy farming scenarios in the Waikato region (Chapter 6).

This required careful consideration of the most appropriate LCA modelling approaches for particular research objectives and research questions.

1.5 Structure of dissertation

In this research, six interlinked steps collectively aiming to appraise the environmental performance of milk production systems in New Zealand were established (see Figure 1.3), starting from a literature study to identify state-of-the art LCA methodologies and knowledge gaps, through to performing a series of LCA studies to: (i) fill the gaps, and

(ii) generate new environmental information and knowledge with regard to applicability of different LCA modelling approaches to address different research questions.

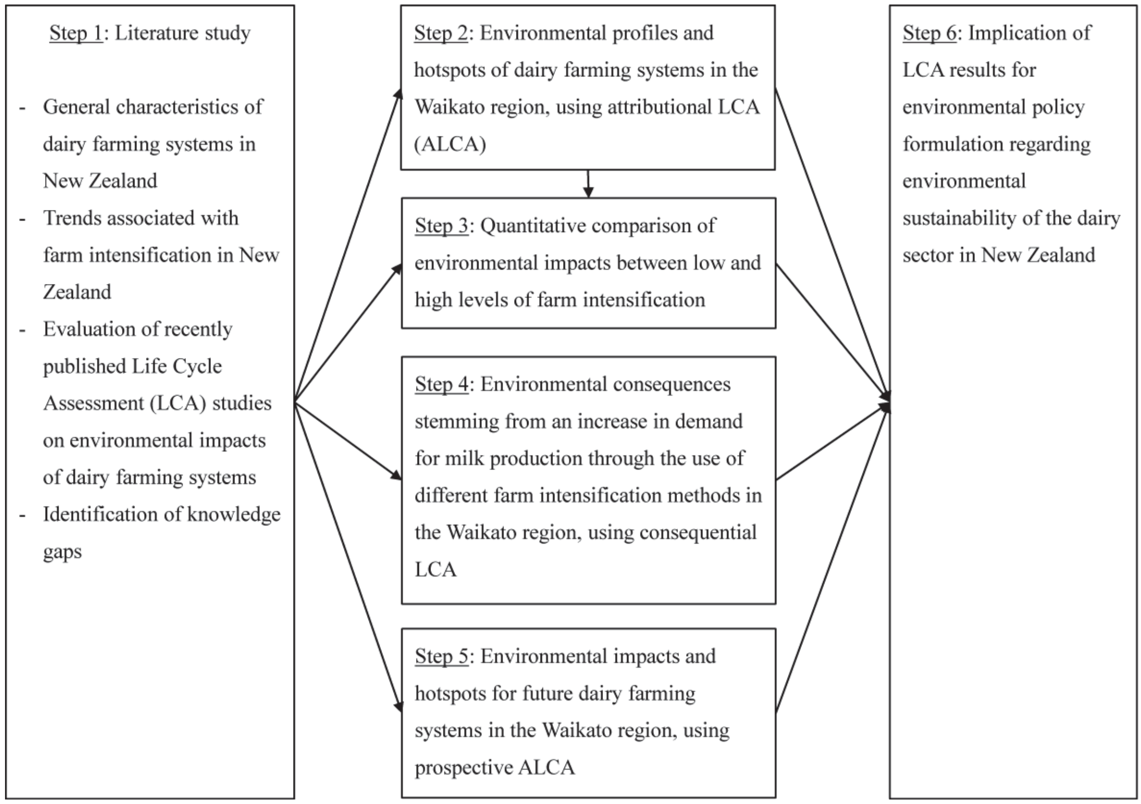

Figure 1.3 The framework for appraisal of environmental sustainability of milk production systems in New Zealand applied in the present study.

The dissertation comprises eight chapters. Chapter 1 introduces the overall rationale, research goals, research questions and the structure of dissertation. In Chapter 2, general characteristics of pasture-based dairy farming systems in New Zealand and their intensification trends, as well as recently-published LCA studies on the cradle-to-farm gate life cycle of milk (i.e. dairy farming systems), are reviewed. In addition, in this chapter, LCA methodological issues and recommendations are evaluated.

Subsequently, the environmental profiles of 53 individual dairy farm systems in the Waikato region were assessed using ALCA, and the results are presented in Chapter 3. These dairy systems were statistically classified into low and high levels of farm intensification, based on determining farm attributes (e.g. stocking rate and levels of brought-in inputs). Multiple environmental impacts between the low and high levels were statistically compared, and the results are presented in Chapter 4.

In Chapter 5, CLCA was used to assess the environmental consequences of an increase in demand for pasture-based milk production through the use of selected farm intensification methods/strategies. In this chapter, identification of marginal suppliers or technologies and competing (displaced) product systems was performed. The identified marginal suppliers and competing product systems were then included in the studied systems. Chapter 6 uses prospective ALCA to assess potential environmental impacts and identify environmental hotspots for a range of possible dairy farming scenarios based on the intensification methods identified in Chapter 5. Chapter 7 discusses the results of these different LCA studies with respect to identification of the environmental hotspots of New Zealand's pasture-based dairy systems, and the implications for use of alternative LCA modelling approaches to support different decision-making situations. Finally, the significance and key findings of this research, as well as recommendations for future research, are presented in Chapter 8.

1.6 Publications arising from the present research

Parts of the dissertation have been modified and submitted/accepted for publication in different peer-reviewed scientific journals as well as national and international conferences, and these are listed below.

Peer-reviewed scientific journal articles:

Chobtang, J., Ledgard, S.F., McLaren, S.J., Zonderland-Thomassen, M. and Donaghy, D.J. 2016. Appraisal of environmental profiles of pasture-based milk production: a case study of dairy farms in the Waikato region, New Zealand. International Journal of Life Cycle Assessment. 21 (3): 311-325 (Chapter 3).

Chobtang, J., Ledgard, S.F., McLaren, S.J. and Donaghy, D.J. 2016. Life cycle environmental impacts of high and low intensification pasture-based milk production systems: a case study of the Waikato region, New Zealand. Journal of Cleaner Production (accepted) (Chapter 4).

Chobtang, J., McLaren, S.J., Ledgard, S.F. and Donaghy, D.J. 2016. Consequential Life Cycle Assessment of pasture-based milk production: a case study in the Waikato region, New Zealand. Journal of Industrial Ecology (accepted) (Chapter 5).

Chobtang, J., McLaren, S.J., Ledgard, S.F. and Donaghy, D.J. 2016. Environmental trade-offs associated with intensification methods in a pasture-based dairy system using prospective attributional Life Cycle Assessment. Journal of Cleaner Production (submitted) (Chapter 6).

International and national conference presentations and papers:

Chobtang, J., Ledgard, S.F., McLaren, S.J. and Donaghy, D.J. 2016. Life cycle environmental impacts of future dairy farming intensification: A comparison of intensified systems based on nitrogen fertiliser versus maize silage. The 29th Annual FLRC Workshop, 9th – 11th February 2016. Palmerston North, New Zealand (poster presentation + full paper).

Ledgard, S, **Chobtang, J.**, Falconer, S. and McLaren, S. 2016. Life cycle assessment of dairy production systems in New Zealand. The 29th Annual FLRC Workshop, 9th – 11th February 2016. Palmerston North, New Zealand (oral presentation + full paper).

Chobtang, J., Ledgard, S.F., McLaren, S.J. and Donaghy, D.J. 2015. Life cycle environmental consequences of increased milk production by dairy farm intensification in New Zealand. The 1st Australian Life Cycle Assessment Conference for Agriculture and Food, 23rd & 24th November 2015, Melbourne, Australia (oral presentation + full paper).

Chobtang, J., Zonderland-Thomassen, M., Ledgard, S., McLaren, S., Brandão, M., and Donaghy, D. 2014. Environmental trade-offs of dairy-farming intensification in New Zealand: case study in the Waikato. The 3rd New Zealand Life Cycle Assessment Conference, 2nd & 3rd September 2014, Wellington, New Zealand (poster presentation + abstract).

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Chapter 2 Traits and Trends in Milk Production Systems in New Zealand, and State-of-the-Art Life Cycle Assessment Studies on Milk Production Systems

Abstract

An increase in demand for dairy products resulting from rising global population and wealth drives farm intensification and usually leads to increased milk productivity (milk yield per hectare). New Zealand, as the single largest dairy exporting nation in the world, has intensified its dairy farming such that increased amounts of off-farm inputs are used to supply additional feed for additional animals. This ongoing system change has led to nutrient excesses (e.g. nitrogen and phosphorus) as a result of unequal nutrient import-export, and exacerbated environmental impacts. However, to support the comprehensive environmental sustainability of dairy farming systems, it is important to account for the wide range of environmental inputs and outputs associated with dairy farming systems and to identify environmental hotspots in order to facilitate mitigation/improvement options. Life Cycle Assessment (LCA) is a system-level analytical approach for assessing comprehensive environmental inputs and outputs and identifying environmental hotspots, as well as identifying any shifts of environmental burdens between impact categories, life cycle stages and relevant businesses. The present literature study highlights that LCA is becoming an important tool to assess the environmental impacts of dairy farming systems. However, there is inconsistency in scope setting, inventory procedures and impact assessment methods, leading to increased difficulties in interpretation and arriving at conclusive outcomes. In addition, most recently-published LCA studies on dairy farming systems have focused mainly on a single impact (i.e. Carbon Footprint) or only a few impact categories. This suggests that there may be a risk associated with burden-shifting between (unstudied) impact categories. Based on the outcomes of this literature review, knowledge gaps as well as methodological issues and recommendations for an LCA study on dairy systems are discussed and summarized.

Keywords: Dairy farming system ● Environmental impact ● Farm intensification ● Life Cycle Assessment (LCA) ● New Zealand

2.1 Introduction

The increase in demand for food, resulting from increasing global population and wealth, is a challenging issue, and is considered as the main driver of agricultural intensification (Byerlee et al. 2014; Tilman and Clark 2014). From a production perspective, intensification of agricultural systems generally leads to increased production intensity (e.g. yield per hectare), and is largely supported by increased use of off-farm inputs (Tilman et al. 2011). However, the increased use of these off-farm inputs (e.g. fossil fuels, chemical fertilisers, pesticides and brought-in feeds) to support agricultural intensification results in increased environmental impacts (Stoate et al. 2009; Tscharntke et al. 2012). In addition, globalization that facilitates the trans-boundary marketing of food products and supporting products (e.g. feed and fertiliser) via import-export usually causes nutrient excesses (especially nitrogen [N] and phosphorus [P]) and subsequently exacerbates localized environmental impacts (Bouwman et al. 2013; Lassaletta et al. 2014; Adrian et al. 2015).

In the dairy sector in particular, intensification of dairy farming systems is generally referred to as an increase in milk yield either per unit of farm inputs (Udo et al. 2011) or per unit of dairy farmland (Crosson et al. 2011), which can be achieved through either increased animal productivity (i.e. increased milk per cow) or increased stocking rate (Beukes et al. 2010; Fariña et al. 2011). These intensification methods are often associated with increased use of off-farm inputs (e.g. N fertilisers and brought-in feeds) (MacLeod and Moller 2006; Pinares-Patiño et al. 2009) and skilled farm management practices (e.g. improved pasture production and utilization efficiency) (Clark et al. 2015). Increased use of off-farm inputs contributes not only to increased milk production per hectare but also to additional environmental impacts (MacLeod and Moller 2006; Basset-Mens et al. 2009; Vitousek et al. 2009). However, the influence of this system change on net environmental impacts has not been comprehensively assessed, and is particularly relevant for dairy farming systems in New Zealand as they are being intensified.

Therefore, this chapter provides an overview of dairy farming systems and intensification in New Zealand (Section 2.2), and an introduction to different modelling approaches in LCA (Section 2.3). This is followed by a review of recently-published LCA studies on dairy farming systems (Section 2.4). LCA methodological issues and the related recommendations for dairy farming systems are addressed in Section 2.5, and Section 2.6 provides a summary of the current level of understanding about a comprehensive range of life cycle environmental impact categories for dairy farming systems.

2.2 Milk production systems in New Zealand

2.2.1 General characteristics

Dairy farming systems differ around the world, but can be classified into two broad categories: (i) confinement-based (i.e. indoor feeding systems), and (ii) pasture-based (i.e. outdoor grazing systems) (Knaus 2015). In the former category, cows are generally kept in barns/housing systems and are fed complete rations or conserved feeds, while in the latter category, cows mainly graze pastures and forage crops, and feed demands of animals are managed to match with feed supplies from these pastures and crops (e.g. through implementing a seasonal calving system), which fluctuate seasonally (Peyraud and Delagarde 2013; Knaus 2015).

In New Zealand, dairy farming systems are pasture-based, where pastures comprise mainly perennial ryegrass (*Lolium perenne* L.), with white clover (*Trifolium repens* L.) as the main legume companion (Moot et al. 2009). Generally, rotational grazing systems are used in order to maximize pasture utilization as well as to maintain pasture quality (McCarthy et al. 2014). However, seasonal fluctuations in pasture production limit forage availability, leading to insufficient feed supply especially during the winter and summer periods when pasture growth rates are low (Rawnsley et al. 2007). These feed gaps are usually filled through the use of off-farm inputs (e.g. brought-in feeds and N fertiliser to increase pasture production) or conserved feeds that are produced on-farm using surplus pasture (Clark et al. 2013). Generally, about 80 – 85% of New Zealand milk is derived from on-farm pastures, while the remainder is from brought-in feeds (Clark et al. 2013).

The Waikato region is the single largest dairy region in New Zealand, accounting for ~24% of total milking cows and ~23% of total milk production in the 2013/14 season (DairyNZ 2014). Other key farm traits of dairy systems in the Waikato region are presented in Table 2.1.

Table 2.1 Key farm traits of dairy farming in the Waikato region compared to the New Zealand national average in the 2013/14 season.

Physical farm traits	Units	Waikato	National	Share (%) [a]
Total dairy farms	#	3,536	11,927	29.6
Total farmland	hectare	395,037	1,716,464	23.0
Total milking cows	#	1,164,661	4,922,806	23.7
Farmland area per farm	hectare	112	144	-
Average heard size	#/farm	329	413	-
Average stocking rate	#/hectare	2.95	2.87	-
Total milksolids production	million kg	415	1,825	22.7
Milksolids production per hectare	kg/hectare	1,051	1,063	-
Milksolids production per cow	kg/cow	356	371	-

[a] The proportion (%) of dairy farming in the Waikato region relative to the national totals. Source: DairyNZ (2014).

2.2.2 Trends in milk production systems in New Zealand

According to the survey carried out by DairyNZ (2014), in the 2013/14 season, there were 11,927 dairy farms in New Zealand (with an average of 413 cows per farm, and 144 hectare per farm). Total milksolids (milk fat plus milk protein) production was 1,825 million kg (with an average of 371 kg per cow, equivalent to 4,196 litres of liquid milk per cow) (see Table 2.1). In New Zealand in the 10 years from 2004/05 season to the 2013/14 season, there were 19% fewer farms, but the total population of dairy cows had increased by 74%, total dairy farmland by 46%, average herd size by 114%, average farmland area per farm by 80% and average stocking rate by 19% (Figure 2.1). These trends resulted in increases in total milksolids production per hectare and per cow by 58% and 37%, respectively, over the same period (see Figure 2.1).

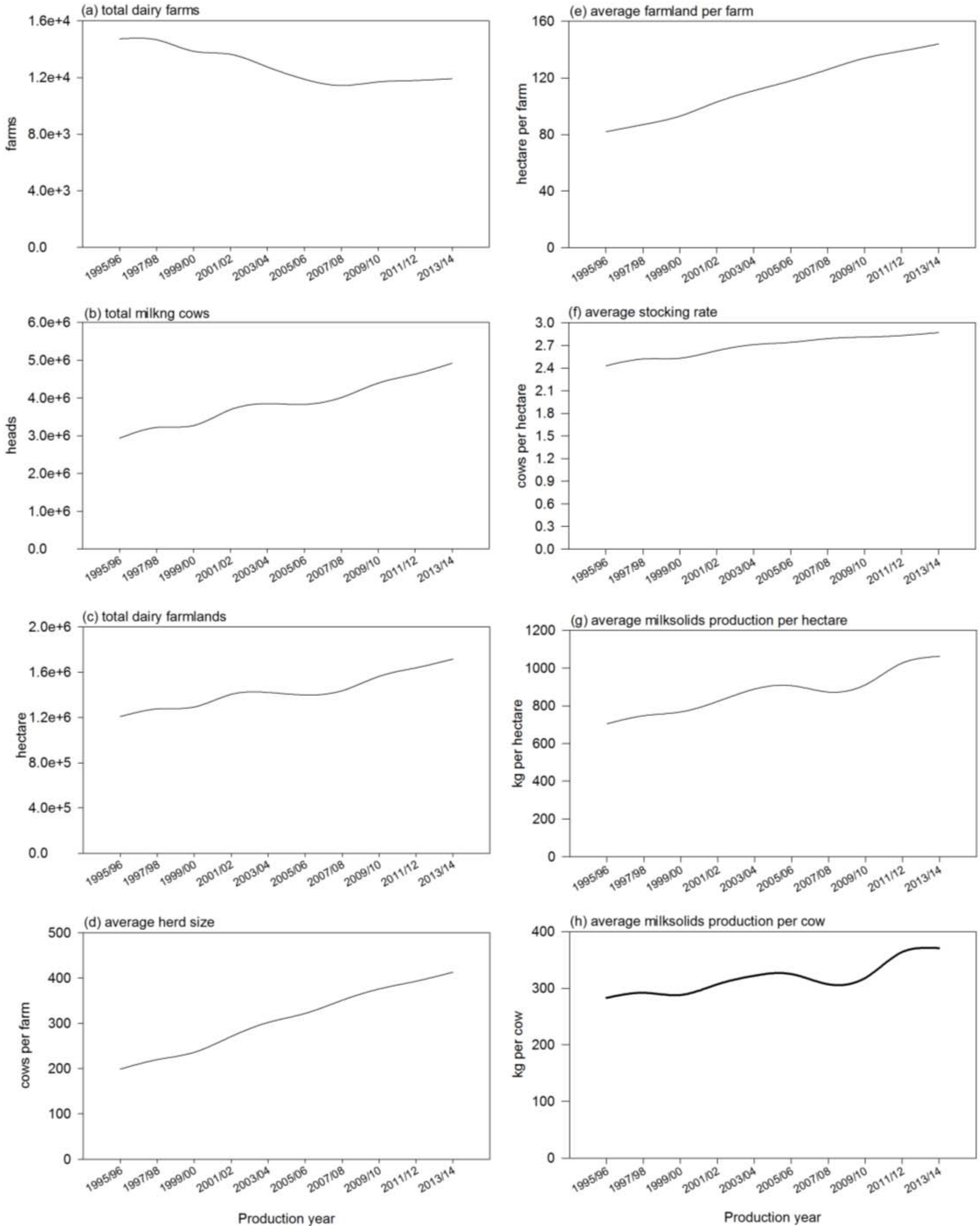

Figure 2.1 Trends in physical farm traits of dairy farming systems in New Zealand over the production years of 2004/05 – 2013/14: (a) total number of farms, (b) total cow population, (c) total dairy farmland, (d) average herd size, (e) average farmland per farm, (f) average stocking rate, (g) average milksolids production per hectare, and (h) average milksolids production per cow. Source: DairyNZ (2014).

In parallel with increasing trends in physical farm traits, trends associated with the total amounts of imported palm kernel expeller (PKE) and N fertilisers were observed. The total amounts of imported PKE and N fertilisers increased by 799% and 68%, respectively, over the decade to 2014 (see Figure 2.2). Note that most PKE is used in dairy systems.

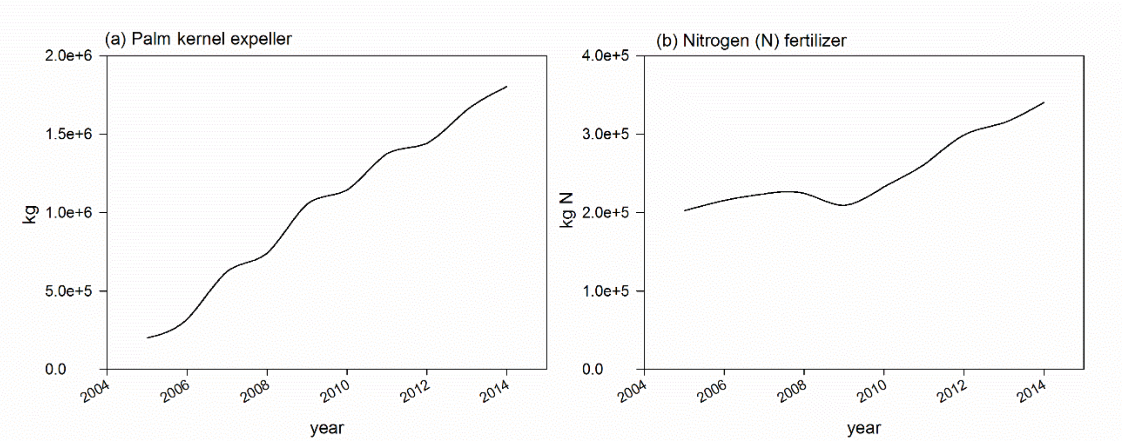

Figure 2.2 Trends associated with total imported quantities of: (a) palm kernel expeller, and (b) nitrogen fertilisers during 2004 – 2014. Source: StatisticsNZ (2015).

2.2.3 Potential intensification options

In 2024, the predicted global demand for dairy products will be ~30% greater than that in 2014 (OECD/FAO 2015). In order to support such an increased demand, the global dairy sector will have to increase its production capacity. In principle, two main strategies can be used to increase total milk production: (i) farm expansion, and (ii) farm intensification (Beukes et al. 2010; Fariña et al. 2011). The former approach requires additional land to be converted either from other existing land-based agricultural systems (e.g. crops and forestry) or nature (e.g. forest and native grassland) to establish new farms. The latter approach keeps the existing area of dairy farmland constant, even though marginal land is inevitably needed to produce off-farm inputs to support farm intensification (e.g. extra cows and feed crops).

For pasture-based dairy systems, farm intensification is generally associated with increased stocking rate (more cows per ha). This can result in a shortage of feed supply from pastures due to increased feed demands by a greater number of animals per hectare. This issue is usually addressed by increased use of off-farm inputs (e.g. brought-in feeds, N fertilisers and freshwater for irrigation) in order to increase feed supply (McDowell et al. 2011; Parfitt et al. 2012; Ledgard and Boyes 2013).

In summary, the common intensification method to increase milk production per hectare in New Zealand is associated with increased stocking rate, coupled with: (i) increased feed supply from on-farm pastures, and/or (ii) increased use of brought-in feeds.

2.2.3.1 Increased pasture production

The ultimate goal of pasture management is to optimize the production, quality and utilization of pasture (Rawnsley et al. 2007). In particular, the improvement in pasture production and utilization is considered one of the key factors supporting the profitability of pasture-based dairy farming operations (Newman and Savage 2009; O'Brien et al. 2015). As mentioned earlier, perennial ryegrass and white clover is extensively used in New Zealand dairy farming systems. This pasture normally yields 10 – 14 t per hectare annually (on a dry matter [DM] basis) with a high N concentration, ranging from 2.9 to 5.1% DM (Lee et al. 2012; Smith et al. 2012). In general, apart from development in genetic potential of the pasture species, an increase in pasture production can be achieved by increased use of N fertilisers and/or irrigation during the year (Ledgard 2006; Pinares-Patiño et al. 2009).

Increased use of nitrogen fertiliser

An increase in milk production is generally associated with the total amount of N fertilisers used (Beukes et al. 2012; Finneran et al. 2012), mainly due to N fertilisers enhancing the growth of pastures. The use of N fertilisers is considered a key technology for dairy farming systems in New Zealand (Pinares-Patiño et al. 2009). Strategic use of N fertilisers to increase pasture production to meet periods of feed shortage or for intensifying dairy farming in New Zealand has been well documented (Ledgard et al. 1998; Ledgard 2006). Planned strategic application of N fertilisers can support their effective and profitable use in dairy farming systems (Ledgard 2006). For example, N fertiliser is used to boost pasture growth prior to the calving period, or to maximize pasture growth during spring months where the surplus pasture is conserved (e.g. making silage or hay) for use during the summer or winter months when there is a shortage of feed from pastures.

Intensification of dairy farming systems in New Zealand through the use of N fertilisers improves yield and quality of pastures, and hence milk production per hectare (Harris et al. 1996; Pembleton et al. 2013). However, apart from environmental emissions associated with manufacturing of N fertilisers (Hasler et al. 2015), other negative effects arising from use of N fertiliser on the environment have also been observed (Ledgard et al. 2009; Beukes et al. 2012). For example, increased use of N fertilisers leads to increased N surplus (Beukes et al. 2012), which generally results in contamination of waterways (Saggar et al. 2011). As a result, net environmental emissions can be significantly increased when the upstream processes (e.g. manufacturing and transport) are accounted for (Ledgard et al. 2011; Hasler et al. 2015). Therefore, this trade-off must be carefully considered.

Irrigation

Irrigation is considered one of the key factors supporting intensification of dairy farming in New Zealand (Pinares-Patiño et al. 2009). In principle, irrigation is targeted mainly at enhancing pasture yield, especially during a summer dry period when the pasture growth is slow (Dalley and Geddes 2012). Perennial ryegrass pasture has poor persistence under dry conditions (Neal et al. 2009; Chapman et al. 2012). With irrigation, the growth rate of the grass is markedly increased, especially in dry land regions. Houlbrooke et al. (2011) found that irrigated paddocks in the North Otago region produced at least twice the annual yield of non-irrigated paddocks. In 2010, the total irrigated area in New Zealand was approximately 972,000 hectares; of this figure, 647,000 hectares (67%) was in the Canterbury region, where dairy farming is the major water user (McDowell et al. 2011).

Although irrigation is a key factor supporting further intensification of the dairy farming systems in New Zealand (Clark et al. 2013), some adverse effects on the quality of pasture soil and freshwater ecosystems as a result of irrigation have been reported (Houlbrooke et al. 2011; McDowell et al. 2011). For example, an increase in N leaching and P run-off through to waterways resulted from the use of irrigation (McDowell et al. 2011). In addition, Houlbrooke et al. (2011) reported that irrigation caused soil compaction, which subsequently increased risks of nutrient (N and P) runoff. Moreover, energy consumption (fossil energy and electricity) in irrigated dairy farming systems is far greater than that in non-irrigated dairy systems (Barnett and Russell 2010). Therefore, it is important to consider these trade-offs in order to maximise benefits, while minimizing negative effects.

2.2.3.2 Increased use of supplementary feed sources

From an animal nutritional perspective, there are limits to increased milk production per cow from grazing pasture alone (Kolver 2003; Clark et al. 2013). Additional milk production must come from extra feeds brought into farms. In New Zealand dairy systems, increased use of brought-in feed supplements is apparent (Figure 2.3) (Ledgard and Boyes 2013). For example, the use of PKE in dairy systems in New Zealand increased more than 700% between the production years 2004/05 and 2010/11 (Ledgard and Boyes 2013). Feed supplements in New Zealand dairy farming can be classified into three broad categories: (i) grazed forage crops, (ii) conserved forage, and (iii) concentrates/by-products (Clark et al. 2007).

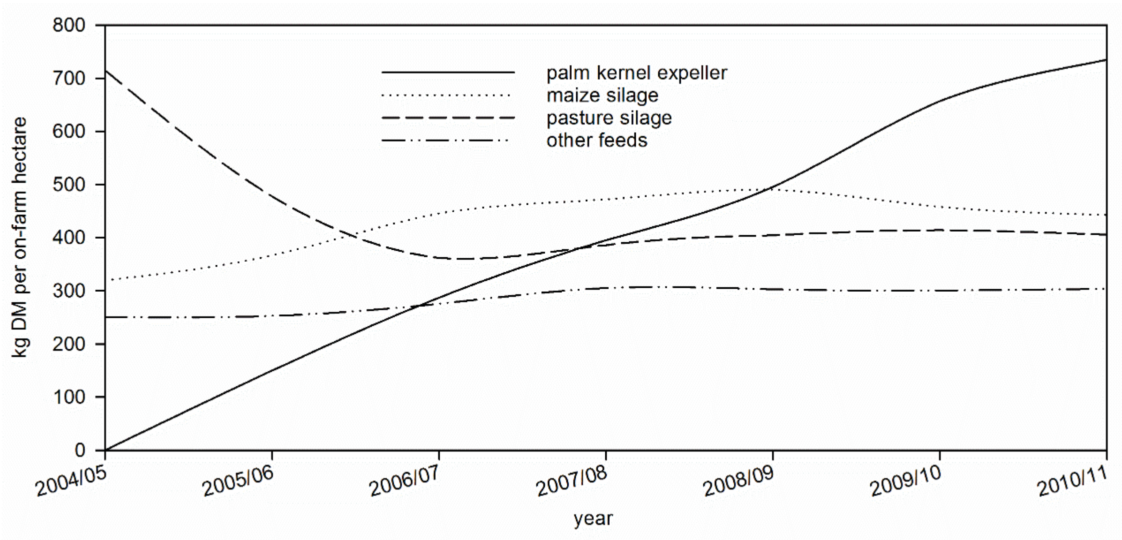

Figure 2.3 Trends associated with amounts of brought-in feeds used on New Zealand dairy farms. Source: Ledgard and Boyes (2013).

Grazed forage crops

Dairy farming intensification through the use of grazed forage crops (e.g. Brassica spp.) has been successfully implemented in New Zealand (Barry 2013). It has been reported that brassica forage is a good source of feed, in particular during the summer and winter months when the growth rate of pastures slows down (Barry 2013). Under New Zealand conditions, brassica crops can yield 6.5 – 14.4 t DM per hectare (Fletcher and Chakwizira 2012). Westwood and Mulcock (2012) compared the nutritive values of five Brassica species, namely leafy turnips, bulb turnips, rape, swede and kale grown as forage under New Zealand climate, and found that these forages had high concentrations of metabolisable energy, ranging from 12.2 to 13.8 MJ per kg DM; in comparison, grazed grass at a similar time had a metabolisable energy concentration of 11.5 MJ per kg DM (Kolver 2003).

Conserved forage

In New Zealand, the growth rate of pastures is uneven throughout the year (Moot et al. 2009), with a peak during the spring months that generally exceeds animal requirements. This surplus can be conserved and used to feed animals when pasture production is insufficient to meet their feed demand (e.g. during summer and winter months). Pasture silage may be produced within the dairy farm but it is also brought-in from outside the farm. Woodward et al. (2006) compared the milk performance of dairy cows with and without pasture silage supplementation during summer, and found that supplemented grazing cows had higher total feed intake and subsequently higher milk production compared with supplemented cows.

Maize is another high yielding forage crop, specifically grown for producing silage, which has increasingly been used to increase milk production and reduce risk from climatic uncertainty in dairy farming systems in New Zealand (Luo et al. 2008; Booker 2009). Maize can yield 20 – 25 t DM per hectare (Densley et al. 2001), however, lower yields (16 – 20 t DM per hectare) have been reported for maize grown in the South Island (Wilson et al. 1994; Millner et al. 2005). In general, maize silage has a moderate nutritive value (Millner et al. 2005), with a metabolisable energy concentration of 10.3 MJ per kg DM, and an N concentration of 1.2% DM (Kolver 2003). In the 2010/11 season, maize silage comprised ~22% of total brought-in feeds used in an average dairy farm in New Zealand (Ledgard and Boyes 2013).

The benefits of using supplementary feeds to increase milk production in dairy farming systems are obvious. However, additional environmental emissions associated with production of these supplementary feeds must be accounted for to ensure net benefits in regard to environmental sustainability.

Concentrates and by-products

Generally, forage-based feeds (e.g. maize silage) may not fully meet the nutrient requirements of dairy cows due to nutritional or feed quality constraints (Kolver 2003). Therefore, concentrates or by-products high in nutrients (e.g. protein and energy) may need to be added in order to reduce nutritional constraints and support animal productivity. For example, Dias et al. (2008) reported that PKE can be given as a supplement to lactating cows in order to extend lactation, resulting in an improvement of farming productivity.

It is important to note that increased use of brought-in feed supplements may cause a large surplus of nutrients (e.g. N and P) at the farm level. Gourley et al. (2012) reported that the risk of nutrient pollution in dairy farming can significantly increase when nutrient inputs exceed the amount of nutrients leaving the farm in product. These surpluses generally lead to environmental problems, e.g. N leaching and P runoff contaminating waterways (Rotz et al. 2005; Gourley and Weaver 2012). Therefore, this issue also needs to be considered in order to improve environmental performances of dairy systems.

2.3 Life Cycle Assessment

As introduced in Chapter 1 (Section 1.3), LCA is a system-level analytical approach that is increasingly being used to quantify a range of environmental impacts, reveal environmental hotspots and identify environmental burden-shifting along the life cycle of a product (Finnveden et al. 2009). This tool has gained increasing attention for its role in informing environmentally-focused sustainable development, including in the dairy sector (FAO 2010). Generally, LCA can prevent problems associated with shifting of environmental burdens from one life cycle stage to other stages, from one impact category to other impact categories or from one business to other businesses (Finnveden et al. 2009; Hellweg and Milà i Canals 2014). Moreover, a contribution analysis can assist in identification of environmental hotspots which subsequently facilitates identification of opportunities to reduce environmental impacts over the life cycle of a product system. The ISO 14040 and 14044 standards (International Organization for Standardization 2006a, 2006b) provide principles and guidelines to undertake an LCA study.

In principle, there are four interlinked methodological phases in LCA (Figure 2.4): (i) Goal and scope definition, (ii) Inventory analysis, (iii) Impact assessment, and (iv) Interpretation (International Organization for Standardization 2006a, 2006b). As they are interlinked, all phases are usually revised or adjusted at any time over a study period in order to ensure that the pre-setting research goal, and research question are properly handled and addressed.

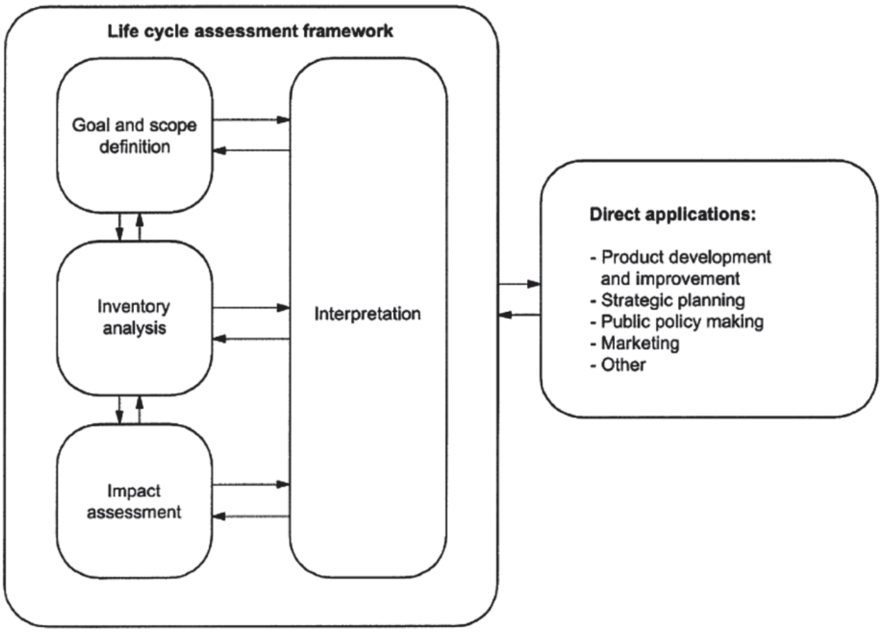

Figure 2.4 A general framework of Life Cycle Assessment, illustrating interlinking among the four life cycle phases. Source: International Organization for Standardization (2006a p. 8).

Even though it is not clearly indicated in the ISO standards, it is widely accepted that there are two schools of thought associated with modelling for LCA: (i) attributional (ALCA) and (ii) consequential (CLCA) (Finnveden et al. 2009; Weidema et al. 2009). These two LCA modelling approaches are different regarding their applicability, intended use, and interpretation of results. The ALCA aims to describe (actual) physical flows (i.e. resources and environmental emissions) to and from the life cycle of an analysed product system (Finnveden et al. 2009). The CLCA focuses on describing how physical flows to and from the technological sphere are changed in response to a change applied in the life cycle of a studied product system (Weidema et al. 2009). In addition, these two LCA modelling approaches are different with respect to their data requirements (e.g. average or marginal) and methods to handle co-products (e.g. allocation or avoided allocation), leading to differences in intended use and result interpretation.

2.3.1 Attributional Life Cycle Assessment

In general, ALCA provides information associated with the environmental impacts resulting from each of unit processes that are included in the life cycle of a product system (Finnveden et al. 2009). It does not consider indirect effects (i.e. environmental effects occurring outside a defined system boundary). Often, ALCA is used to assist in identification of opportunities for reducing direct impacts in different parts of a product's life cycle, and usually uses average data for each of the unit processes. It is important to note that ALCA assesses environmental impacts of a product system based on a status quo or static view, meaning that all unit processes involved in the product system being studied are fixed, implying that any changes (increases or decreases) are not a focus of the analysis. In the case where a unit process requires or produces more than one product (i.e. co-products), resource use and environmental emissions are partitioned or allocated to each of the co-products based on specific relationships, e.g. mass and economic values.

2.3.2 Consequential Life Cycle Assessment

Theoretically, CLCA provides information associated with the consequences of a change in the level of production or a change in demand for a studied product (functional unit). It can account for environmental effects derived from both inside and outside the boundary of a product system being studied (Weidema et al. 2009). In other words, CLCA models a causal relationship originating from a decision to change the output quantity of a studied product. Basically, CLCA has two specific characteristics: (i) it models the technology (unit process) that is actually affected by a change in demand for a product (marginal technology), and (ii) it handles co-products by avoiding allocation (Weidema et al. 2009).

The marginal technology (or marginal supplier) refers to the technology whose production capacity can be adjusted in response to a decision or a change in demand. Further, the preferred marginal technology can be identified by the production cost per unit. Thus, the technology that can provide the lowest costs per unit of products (based on a long-term perspective) is likely to be the most preferable marginal technology (Ekvall and Weidema 2004; Schmidt 2008). Regarding the co-product handling method in CLCA, it is compulsory to avoid allocation by using system division, system expansion or system substitution (Weidema et al. 2009; Pelletier et al. 2015). The main benefit of avoiding allocation is that it can maintain a balance in all aspects (e.g. mass, element, and economic) between inputs and outputs of an analysed system (Weidema and Schmidt 2010; Pelletier et al. 2015).

For environmental studies on dairy farming systems, at the time of the present literature study, there were only two published papers using CLCA. First, Thomassen et al. (2008) modelled the consequences of increased demand for milk, assuming that one more farm was established. These authors identified marginal suppliers using a stepwise method (Ekvall and Weidema 2004). The marginal suppliers considered were for supply of electricity, fodder protein and fodder energy. The authors handled co-products using system expansion. In addition, different competing products (i.e. products that were identified to have similar functions or market segments to the studied co-products, e.g. conventional beef (or pork) to be replaced (substituted) by the co-product of dairy meat) were tested.

Second, Nguyen et al. (2013a) delimited their study based on Thomassen et al. (2008) and used CLCA to assess environmental impacts associated with a dairy system change from maize silage-based to pasture-based in France. These authors assumed that the levels of milk production between pre- and post-system change were unaffected, and took into account effects associated with indirect land use change (ILUC; i.e. any extra demand for land occurring as a result of displacement of existing land use by an extra demand for supporting products, e.g. feeds) on net environmental impacts of milk. Other consequences of this system change (e.g. a reduction in demand for soybean meal and an increase in a demand for wheat and rapeseed in the pasture-based system) were also accounted for.

2.4 Life Cycle Assessment studies on milk production systems

For this review, scientific papers published over a period of four years (2011 – 2014) were included. The papers chosen were based primarily on a cradle-to-farm gate perspective (i.e. liquid milk production at the dairy farm gate was a focus), although additional studies that used different system boundaries were also considered for the purpose of comparison. As a result, 21 papers were evaluated (see Tables A1 and A2 of the Supplementary material A for a list of the sourced papers and summarised details). It should be noted that almost all of the older dairy LCA scientific papers (published before 2011) have been reviewed by de Vries and de Boer (2010) and Yan et al. (2011). Therefore these papers were not discussed in this review except for those that covered issues not addressed by the two previous review papers (e.g. impacts associated with

ILUC and some specific impact categories such as Ozone Depletion Potential [ODP] and Photochemical Ozone Formation Potential [POFP]).

The papers reviewed had different objectives and used different functional units, depending mainly on the specific objectives of the studies. Most of the papers reviewed used different life cycle inventory models and Life Cycle Impact Assessment methods, except for the impact on Climate Change (CC) where the characterisation factors derived from the Intergovernmental Panel on Climate Change (2007) were used. In addition, the majority of the papers reviewed assessed environmental impacts for confinement-based systems in Europe (15 papers); only a few papers reported environmental impacts of different dairy farming systems in North America (3 papers), Oceania (2 papers) and South America (1 paper). All the papers reported the CC impact but only 10 papers included a few more impact categories, including Acidification Potential (AP) and Eutrophication Potential (EP). It is important to note that there is no paper that assesses a comprehensive range of environmental impacts associated with use of different farm intensification options to increased milk production.

2.5 Methodological issues and recommendations

Life Cycle Assessment is increasingly becoming an important approach to assess environmental impacts and identify environmental hotspots in the life cycle of raw milk as well as other dairy products (International Dairy Federation 2010). However, there are always controversial issues associated with choice and applicability of LCA methodology. These issues usually result in inconsistent methodologies applied across LCA studies, subsequently leading to inconclusive and incomparable results between studies.

This section provides a review of current methodological issues and recommendations for LCA studies on dairy farming systems. The ISO 14040 and 14044 standards (International Organization for Standardization 2006a, 2006b) were used as a “baseline” for evaluation of the methodological issues. Specific LCA guidelines for the carbon footprint (CF) of dairy products (International Dairy Federation 2010) and LCA guidelines for the Product Environmental Footprint (European Commission 2013) were also considered and referred to where relevant.

2.5.1 Goal definition

General description

Defining the goal or objective is the first phase in LCA studies. The goal of LCA studies usually determines subsequent steps (phases) and defines methodological choices, e.g. modelling approach, system boundary and functional unit (International Organization for Standardization 2006b).

Most reviewed studies have set the study goals either as: (i) quantification of potential environmental impacts, or (ii) identification of environmental hotspots. Some of studies have compared the impacts between different dairy systems, different co-product handling methods and different mitigation options.

Methodological issues

Even though there is no detailed guidance that describes how to define goal in LCA analyses, the International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC-IES 2010a) indicates that it is necessary to identify whether the defined goal is supporting decision-making, and link this situation to appropriate modelling approaches to be used. In the case where LCA supports decision-making and such decision-making has consequences of different sizes (e.g. small or large), the Handbook recommends using ALCA for a small consequence and CLCA for a large consequence. For an LCA study that does not support decision-making but aims to monitor the environmental performance of a studied product system, the Handbook recommends using ALCA. However, the ILCD Handbook has recently been challenged by Ekvall et al. (2016) regarding the appropriateness of its recommendations.

Methodological recommendations

Setting the research goal is one of the most important phase in LCA studies because it shows a reason for carrying out the study. In addition, the study goal determines the subsequent procedures (choices) to be used in order to achieve such a goal. The research goal generally determines the research question. As well, a defined research question assists in selection of appropriate LCA modelling, e.g. either ALCA or CLCA. For example, where the goal is to assess potential environmental impacts and/or environmental hotspots in a fixed dairy system (i.e. a status quo), ALCA is recommended. However, when the goal is to investigate potential environmental impacts associated with a change (either increase or decrease) in demand for milk production or to compare potential environmental impacts resulting from dairy system changes, CLCA is recommended. However, the recommendation associated with appropriate LCA modelling to support decision-making remains a question.

2.5.2 System boundary

General description

The system boundary in LCA defines inclusion or exclusion of unit processes in an analysed product system (International Organization for Standardization 2006b). The system boundary can be drawn at various places, depending on the goal of the study, e.g. cradle-to-farm gate or cradle-to-grave. For most LCA studies reviewed where environmental impacts associated with the life cycle of raw milk production were considered, the cradle-to-farm gate system boundary is normally chosen, meaning that all resource use and emissions occurring in the life cycle of raw milk production, starting from acquisition of raw materials through to milk at the dairy farm gate are accounted for. Basically, this system boundary covers 60 – 80% of most environmental impacts of the entire supply chain of dairy products (Meneses et al. 2012; Thoma et al. 2013b).

For processed dairy products (e.g. drinking milk, milk powder, butter and cheese), a cradle-to-grave perspective is often used as a system boundary. For example, Thoma et al. (2013b) assessed environmental impacts associated with the life cycle of drinking milk in the USA, Kim et al. (2013) assessed environmental performances in the life cycle of cheese and whey in the USA and González-García et al. (2013) quantified environmental impacts associated with the life cycle of yoghurt in Portugal.

Methodological issues

Even though most flows in the cradle-to-farm gate life cycle of milk were included in all studies reviewed, some flows (e.g. capital goods, and veterinary material and services) were excluded. Most reviewed studies indicated that inventoried data for these flows were not available, and assumed the contribution of these flows were insignificant. However, it has been reported that the contribution of capital goods can be significant for some impact categories, e.g. Human Health Toxicity and Ecotoxicity for Aquatic Freshwater (Ecotox) (Frischknecht et al. 2007; Blengini and Busto 2009).

Roer et al. (2013) used ALCA to quantify environmental impacts of a milk production system in Norway. These authors accounted for the emissions arising from production and use of farm buildings and machinery as well as fencing materials. Their result showed that the impacts on Climate Change (CC) and Marine Ecotoxicity Potential (MEP) (on a per kg milk) increased by 13% and 34%, respectively when emissions from these capital goods were accounted for, and the main contributors to these increases were the manufacturing of agricultural machinery and buildings, respectively.

Methodological recommendations

For LCA studies on the life cycle of raw milk production systems, a cradle-to-farm gate perspective (the so-called partial LCA) is recommended. For processed dairy products, a cradle-to-grave perspective is recommended in order to cover all resource use and environmental emissions over the entire supply chain. The European Commission (2013) recommended that the emissions arising from capital goods and their relevant use should be accounted for by applying a linear model to calculate depreciation and allocate environmental impacts.

2.5.3 Functional unit

General description

The functional unit (FU) is defined as the quantified performance of a product system for use as a reference unit to which the environmental impacts are expressed (International Organization for Standardization 2006b). The FU is very important because all environmental emissions to be determined are related to it (Finnveden et al. 2009), and it is generally formulated in accordance with the goal of the study.

Methodological issues

The main function of dairy farming systems is to produce milk. Therefore, the milk produced is suitable to be used as the FU of dairy farming systems, although other aspects can be used when there is a different study goal. For example, a hectare of farmland can be used as a FU for the life cycle of dairy farming systems when land productivity is intended to be evaluated and compared. However, most reviewed LCA studies used a unit of milk produced as a FU. A unit of standardized milk production is often used as it facilitates a fair comparison between LCA studies, even though some non-standardized units are sometimes used, e.g. 1 kg and 1 L of raw milk (Verge et al. 2013).

Methodological recommendations

Standardized milk (fat- and protein-corrected milk, FPCM) is recommended as a FU for the cradle-to-farm gate life cycle of dairy farming systems (FAO 2010; International Dairy Federation 2010). This FU ensures a fair comparison between different dairy farming systems where fat and protein concentrations may vary (Bertrand and Barnett 2011). The formulae to calculate FPCM containing 4.0% fat and 3.3% protein are as follows.

- FPCM (kg) = raw milk (kg) × [0.337 + (0.116 × % fat) + (0.06 × % crude protein)] (FAO 2010).
- FPCM (kg) = raw milk (kg) × [0.2534 + (0.1226 × % fat) + (0.0776 × % true protein)] (International Dairy Federation 2010).

2.5.4 Co-product handling method

General description

A multifunctional process (referred to as the flow or process that has either more than one valuable or usable inputs, or outputs) can be either inflow or outflow, and is generally a critical issue in LCA studies (Finnveden et al. 2009; Guinée et al. 2009). A method is needed to handle and assign environmental burdens to the co-products in a multifunctional process, and the outcomes of LCA studies are highly sensitive to the co-product handling methods (Gnansounou et al. 2009; Kaufman et al. 2010; Wardenaar et al. 2012), including dairy products (Feitz et al. 2007; Flysjö et al. 2011a). Therefore, it is important to use appropriate co-product handling methods (Finnveden et al. 2009; Guinée et al. 2009).

Methodological issues

According to guidelines of the International Organization for Standardization (2006b), the stepwise (hierarchical) approach is recommended for handling co-products in LCA studies. First, avoiding allocation by using either system subdivision, system expansion or system substitution is the most preferable method. Second, where allocation cannot be avoided, environmental burdens should be partitioned (allocated) based on an underlying physical relationship, emphasising that this relationship must reflect the way in which inputs and outputs are changed by quantitative changes in the products or functions delivered by the system. Finally, in a case where a physical relationship cannot be applied, allocation should reflect other relationships between inputs and outputs of the system, such as economic values. For CLCA, it is recommended that co-products must be handled by avoiding allocation through the use of system expansion or system substitution since this models environmental impacts resulting from a change in demand for functional unit that can influence (affect) the interconnected product systems (Weidema et al. 2009; Pelletier et al. 2015).

The ISO 14044 standard (International Organization for Standardization 2006b) recommends that co-product handling methods should be consistent throughout the system being studied, implying that only one co-product handling method should be applied over the entire product system being assessed. However, it can always be observed that, in the life cycle of dairy farming systems, different allocation methods have been applied between co-products of inflow (e.g. feed ingredients derived from food and bioenergy systems) and outflow (e.g. milk and meat). This is due to the fact that there are limitations imposed by the availability of relevant data and extremely different functions of the co-products (e.g. oil palm [food or energy] and palm kernel expeller [feed]), especially for feed ingredients where there are a very large number of potential feed products that are derived from different production systems.

Methodological recommendations

According to the stepwise (hierarchical) procedure outlined by the ISO 14044 standard (International Organization for Standardization 2006b), it is recommended that avoiding allocation through using system expansion (substitution) is the most preferable choice to handle co-products in LCA studies on dairy farming systems. However, this method should be applied only for CLCA, which models a change in demand (i.e. milk production) rather than for ALCA (Thomassen et al. 2008; Brander and Wylie 2011; Pelletier et al. 2015). It is important to define the relevant assumptions and identify appropriate competing product systems when using system substitution. It is also recommended that competing product(s) must have similar functions and/or similar market segments as those for the co-product(s) being handled (Ekvall and Weidema 2004; Thomassen et al. 2008).

For ALCA, an allocation method based on the causal biophysical relationship is the most preferable choice to handle the co-products milk and dairy meat (International Dairy Federation 2010; Thoma et al. 2013a), whereas an economic relationship is recommended for handling co-products associated with inflow (International Dairy Federation 2010).

2.5.5 Land use and land use change

Like other agricultural systems, land use in dairy production systems can be either a source or a sink in the carbon cycle. An increase in soil carbon results in removal of carbon dioxide (CO_2) from the atmosphere. In contrast, a decrease in soil carbon indicates greater CO_2 emissions to the atmosphere (van Middelaar et al. 2013). With regard to pasture-based milk production systems, carbon sequestration potentials in temperate pastures have been documented (Soussana et al. 2010). Soussana et al. (2010) reported that decay of accumulated plant litter and roots results in potential carbon sequestration. Moreover, application of animal manure or other soil amendments can increase soil carbon (DeLonge et al. 2013; Taube et al. 2014).

From an LCA perspective, there are two aspects to be considered when assessing land use: (i) land occupation, and (ii) land transformation (Milà i Canals et al. 2007b).

2.5.5.1 Land occupation (land use)

Land occupation or land use (LU) refers to an activity (e.g. crop cultivation) being carried out on an area of land (e.g. hectare) for a period of time (e.g. year). This activity contributes to environmental impacts at different magnitudes, depending on a range of factors, e.g. land condition prior to being used and activities during a period of occupying land, etc.

Methodological issues

Note that the contribution of land use to environmental impacts (e.g. the CC indicator) varies, and a consensual method to account for impacts resulting from the use of land does not yet exist.

Rotz et al. (2010) conducted an LCA study to quantify the CF of milk in the USA, taking into account application of animal manure to crop production (to produce feed for animals) under dairy farming systems. Their results showed that the short-term carbon storage in agricultural land accounted for 0.30 kg CO_2 eq. (carbon dioxide equivalent) per kg ECM (energy corrected milk), lowering the CF of milk from 0.99 to 0.69 kg CO_2 eq. per kg ECM (Rotz et al. 2010). Likewise, Schönbach et al. (2012) compared the CF of milk from a pasture-based production system with a confinement-based system in Austria. They found that the pasture-based system had a 48% lower milk CF compared with the confinement-based system when soil carbon sequestration was taken into account.

Del Prado et al. (2013) quantified greenhouse gas (GHG) emissions of milk production systems in Spain, taking carbon cycles into account. Their results showed that each 100 kg carbon per hectare per year sequestered resulted in a 3% reduction in the milk CF. Belflower et al. (2012) compared GHG emissions of milk between a pasture-based system and a confinement-based system in the USA, taking the effects of carbon sequestration into account. Their results showed that carbon sequestration in the pasture-based system reduced total GHG emissions by 12% compared with the confinement-based system.

Although a rise in carbon storage in soils has a significant potential to reduce the CF of agricultural products, the inherent complexities of soil science are the most important difficulties to be addressed (Pawelzik et al. 2013). Ortiz et al. (2013) reported that spatial variability is one of the main sources of uncertainty with regard to soil carbon stocks and changes. Therefore, there is a need to decrease these uncertainties. Brandão et al. (2011) and Müller-Wenk and Brandão (2010) highlighted the need for site-specific modelling of soil carbon changes in LCA studies in order to reduce uncertainty.

Methodological recommendations

As previously mentioned, consensual methods to account for impacts associated with land use do not exist. However, it has been recommended that impacts associated with LU should be included, but presented separately (International Dairy Federation 2010; Ahlgren et al. 2015).

2.5.5.2 Land transformation (land use change)

Land transformation or land use change (LUC) in agricultural production systems can be classified into two categories: (i) direct and (ii) indirect LUC. Direct land use change (DLUC) can occur when either virgin natural lands (such as forests and native grasslands) or in-use cultivated lands (such as livestock and crop farmlands) are converted to other land use purposes. In contrast, ILUC refers to the market force-driven LUCs, occurring as a consequence of displacement of land use elsewhere in the world (Meyfroidt et al. 2013), which is generally induced by increasing demands for feed, fuel and fibre. The following sections provide insights into DLUC and ILUC issues in the life cycle of dairy farming systems.

Direct land use change

General description

This activity refers to as a situation where a studied product system directly displaces other land-based systems (Ahlgren et al. 2015). A change in farming practices can result in either a source or sink of environmental emissions, depending on a range of factors, as above mentioned for LU.

Methodological issues

The magnitude of GHG emissions associated with DLUC varies, depending mainly on the locations and types of land to be converted. For example, Kirschbaum et al. (2013) stated that a shift in land use from forest to agricultural land can cause a substantial amount of GHG emissions in New Zealand. The CF of beef systems in the Legal Amazon Region can be as high as 700 kg CO_2 eq. per kg carcass when emissions associated with DLUC from newly deforested lands were taken into account (Cederberg et al. 2011). In contrast, converting arable lands (croplands) to pastureland can potentially result in an increase in soil carbon stocks (Soussana et al. 2010).

Methodological recommendations

It has been recommended that all emissions associated with DLUC occurring after the 1st January 1990 should be included in the impact assessment of agricultural products, using five per cent of total emissions arising from the DLUC for calculating the GHG emissions of the products in each year over the 20 years following the change in land use (International Dairy Federation 2010). Using this method, Hutchings and Ledgard (2009) showed that LUC due to conversion of forest to dairy farm systems in New Zealand increased CO_2 emissions of milk production by 33%.

Indirect land use change

General description

Generally, ILUC occurs as an undesignated consequence of land use decisions elsewhere. In other words, ILUC exists when a certain land transformation induces changes outside the system boundary of an analysed product system. Environmental emissions for a specific product arising from ILUC are not as easy to calculate, because it is nearly impossible to specify which LUC is a result of the system investigated (Warner et al. 2014). The calculation is generally performed using specific assumptions or economic models, e.g. general equilibrium models (the whole world economy) and partial equilibrium models (linear relationships among prices, demand and production in the agricultural market). It has been indicated that ILUC must be considered as an aspect under consequential modelling (EC-JRC-IES 2010a).

Methodological issues

Few studies have taken ILUC into account in quantifying GHG emissions in dairy farming systems. One example can be seen from the study of Nguyen et al. (2013a) who compared two different modelling scenarios of ILUC in the life cycle of milk. Their findings revealed substantial differences in GHG emissions of milk when emissions associated with ILUC were accounted for (0.95 kg CO_2 eq. per kg of FPCM without ILUC and 1.95 kg CO_2 eq. per kg of FPCM with ILUC). Flysjö et al. (2012) also quantified GHG emissions in the life cycle of milk, taking the effects of ILUC into account. They showed changes in the final results, depending on the ILUC models used (e.g. 0.52 kg CO_2 eq. per kg of ECM without ILUC and 0.95 kg CO_2 eq. per kg of ECM with the worst case of ILUC).

Hörtenhuber et al. (2010) compared GHG emissions from two different Austrian milk production systems (conventional and organic). The authors took the effects of ILUC into account. They estimated the effects of ILUC by modelling use of soybean meal which is imported from South America, and rapeseed cake imported from Eastern and Central Europe. Their results showed that about 8% of total GHG emissions originated from the effects of ILUC, which resulted predominately from soybean meal production (93%).

The contribution of ILUC can affect the impact on the CF of feeds (Castanheira and Freire 2013; van Middelaar et al. 2013). For example, the CF of PKE increased by 877% when accounting for emissions from ILUC (van Middelaar et al. 2013). Likewise, Castanheira and Freire (2013) found that GHG emissions per kg soybean increased by ~2,500% when emissions resulting from conversion of rainforest to soybean systems were accounted for. It can be seen that the final LCA results associated with ILUC vary greatly and depend largely on the underlining assumptions and modelling approaches of ILUC. The assumptions for accounting for ILUC are usually associated with: (i) the area (size) of ILUC, (ii) the land use prior to ILUC, (iii) the duration of land use (which is associated with soil carbon changes), and (iv) the location of ILUC (Castanheira and Freire 2013).

Methodological recommendations

Generally, there is no agreement associated with measures to account for the impacts associated with LU and LUC (i.e. DLUC and ILUC) across LCA communities. However, the International Dairy Federation (2010) and the European Commission (2013) both recommend quantifying the impacts associated with both LU and LUC. Nevertheless, because high variation associated with underlining assumptions and accounting methodologies can lead to highly uncertain results, the results of LU and LUC must be presented separately. In addition, since ILUC occurs as a result of the displacement chain associated with a change in demand for land, the impacts associated with ILUC should be modelled using consequential modelling (EC-JRC-IES 2010a). Ahlgren et al. (2015) reported that environmental impacts associated with ILUC should be assessed, at least in a sensitivity analysis.

2.5.6 Life Cycle Impact Assessment

General description

Life Cycle Impact Assessment (LCIA) transforms complicated results quantified in the Life Cycle Inventory (LCI) into meaningful figures/indicators. The LCI results are assigned to particular impact categories based on a common cause-effect chain[2] for each of these categories, using characterization models (International Organization for

[2] The outline of the mechanism that encompasses all relevant pathways (i.e. physical, chemical and biological processes), and links the results from the LCI phase to environmental indicators.

Standardization 2006b). In the other words, the main objective of LCIA is to gain insights into the potential environmental impacts based on the results from the LCI phase, under the goal and scope of a study which has been designed at an earlier phase.

Theoretically, the results from the LCI phase will be summarized into fewer environmental categories according to the types of the impacts which will be reflected in the "areas of protection"[3], i.e. human health, natural environment and natural resources (EC-JRC-IES 2010a).

There are two schools of thought with respect to impact characterization approaches: midpoint and endpoint (Finnveden et al. 2009; Hauschild et al. 2013). The midpoint approach is a problem-oriented approach where the LCA results are presented as a series of environmental impact categories (EC-JRC-IES 2010a). The endpoint impact translates the LCI results into impacts on the areas of protection (EC-JRC-IES 2010a). Therefore, these two approaches are partly inter-related as the input data for calculating impacts are basically derived from the LCI. However, they are calculated using different characterization models or factors, which can lead to substantially different interpretations and levels of certainty. Figure 2.5 depicts simplified cause-effect chains to link each of the inventory results to midpoint and endpoint indicators.

[3] The entities that humans want to protect: human health, natural environment and natural resources. The area of protection of human health refers to changes in both mortality and morbidity that are associated with goods or services. The area of protection of natural environment encompasses natural ecosystems in terms of their function and structure, and refers to the negative effects on the function and structure of natural ecosystems as a consequence of exposure to chemicals or physical interventions. The area of protection of natural resources refers to efforts needed to safeguard the availability of resources that can be used.

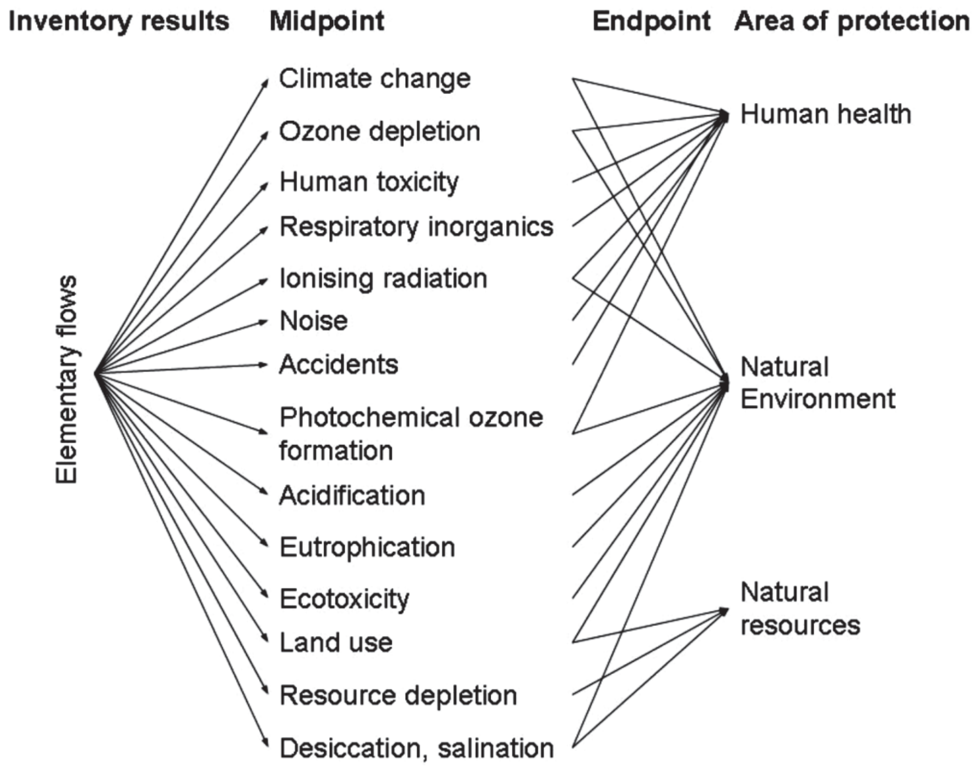

Figure 2.5 Simplified cause-effect chains in Life Cycle Impact Assessment, linking elementary flows derived from Life Cycle Inventory to midpoint and endpoint indicators through to three areas of protection. Source: EC-JRC-IES (2010b p. 3).

Midpoint approach

The impact categories at the midpoint level can be defined by the measurable indicators of potential environmental impacts at any place between the LCI results and the endpoint (Jolliet et al. 2004; Finnveden et al. 2009; Hauschild et al. 2013). At the midpoint level, outputs derived from LCI are aggregated into the impact categories according to the common mechanisms in specified cause-effect chains (Cavalett et al. 2013). A set of default impact categories in this approach has recently been recommended in the ILCD Handbook (EC-JRC-IES 2010a). However, in the LCA studies on dairy farming systems, not all of these impact categories are always assessed. In the case where default impact categories cannot be applied, the European Commission (2013) recommends that the selection of the impact categories for a particular product should be undertaken in a manner that can cover all relevant environmental issues. Moreover, additional impact categories (e.g. information on energy consumption and information on local-specific environmental impacts) that are related to the goals of the studies can be added as complementary to the default impact categories if necessary (European Commission 2013).

Endpoint approach

The impact categories at the endpoint level are generally defined at the level of the areas of protection, i.e. human health, natural environment and natural resources (Hauschild et al. 2013). At this level, the impact indicators, usually derived from the midpoint categories, can be modelled to provide values representing environmental damage at the end of the cause-effect chains. Thus, this approach can basically be carried out by extending the midpoint approach (Cavalett et al. 2013). This approach requires additional models for weighting the results from the midpoint impact categories to the results for the areas of protection (Finnveden et al. 2009).

Recently, both midpoint and endpoint approaches have been developed to support each other in a consistent impact assessment framework (Goedkoop et al. 2012). However, they are finally recalculated using different characterization models or factors which lead them to different interpretation. A summary of the potential benefits of the midpoint and endpoint approaches, modified based on Bare et al. (2000), relating to the topics of robustness and certainty, transparency and reproducibility, comprehensiveness, environmental relevance and relationship with decision support, are presented in Table 2.2. It should be noted that, at the time of this literature study, no published studies associated with the LCA of milk using the endpoint approach have been found.

Table 2.2 Comparative advantages of midpoint and endpoint approaches based on designated criteria according to Bare et al. (2000).

Items	Midpoint	Endpoint
Robustness and certainty	+ + +	+
Transparency and reproducibility	+	+
Comprehensiveness	+	+ +
Environmental relevance	+ +	+
Relationship with decision support	+	+ ++

+ + +; high, + +; medium, +; low

Methodological issues

In general, choice associated with LCIA methods might be significant sources of the difference in the final results of LCA studies (Cavalett et al. 2013; Owsianiak et al. 2014; Bueno et al. 2015). The reviewed studies used different characterization models. Therefore, this generally hampers comparability between these studies.

For LCA studies that focussed only on CF or Global Warming Potential (GWP), most of them used the characterization factors derived from the Intergovernmental Panel on Climate Change (2007). In contrast, for studies that assessed multiple impact categories, some of them (e.g. Bartl et al. 2011; O'Brien et al. 2012) used the CML (Centre of Environmental Science of Leiden University) baseline method (Guinée et al. 2002), whereas the others (e.g. Iribarren et al. 2011; Roer et al. 2013; Yan et al. 2013b) deployed the ReCiPe midpoint method (Goedkoop et al. 2012).

Methodological recommendations

The best existing impact assessment methods have recently been identified using relevant criteria (EC-JRC-IES 2011; Hauschild et al. 2013). Most of those identified impact assessment methods (i.e. impact categories) at the midpoint level have also been accepted by the European Food SCP Roundtable (2013) whose focus is to improve environmental sustainability indicators for the food and drink product sector. These midpoint impact categories are presented in Table 2.3. In contrast, impact categories at the endpoint level have not yet been recommended for use because they remained highly uncertain (EC-JRC-IES 2011; Hauschild et al. 2013).

Table 2.3 A summary of midpoint impact categories recommended by the European Food SCP Roundtable (2013).

Impact category	Units *	Source
Climate Change	kg CO_2 equivalent	Intergovernmental Panel on Climate Change (2007)
Ozone Depletion Potential	kg CFC-11 equivalent	World Meteorological Organization (1999)
Human Health Toxicity – non-cancer effects	CTUh (comparative toxic unit for humans)	Rosenbaum et al. (2008)
Human Health Toxicity – cancer effects	CTUh (comparative toxic unit for humans)	Rosenbaum et al. (2008)
Particulate Matter	kg $PM_{2.5}$ equivalent	Humbert (2009)
Ionising Radiation – human health effects	kg U^{235} equivalent	Dreicer et al. (1995)
Photochemical Ozone Formation Potential	kg NMVOC equivalent	van Zelm et al. (2008)
Acidification Potential	molc H^+ equivalent	Posch et al. (2008) Seppälä et al. (2006)
Terrestrial Eutrophication Potential	molc N equivalent	Posch et al. (2008) Seppälä et al. (2006)
Freshwater Eutrophication Potential	kg P equivalent	Struijs et al. (2009)
Marine Eutrophication Potential	kg N equivalent	Struijs et al. (2009)
Ecotoxicity for Aquatic Freshwater	CTUe (comparative toxic unit for ecosystems)	Rosenbaum et al. (2008)
Resource Depletion – water	m^3 water use related to local scarcity of water	Ridoutt and Pfister (2010)
Resource Depletion – mineral and fossil	kg antimony (Sb) equivalent	van Oers et al. (2002)
Land Use Impact	kg carbon deficit	Milà i Canals et al. (2007b)

* CO_2 = carbon dioxide; CFC-11 = trichlorofluoro-methane; CTU_h = comparative toxic unit for humans; $PM_{2.5}$ = particulate matter less than 2.5 μ in diameter; kBq = kilobecquerel; U^{235} = uranium-235; NMVOC = non-methane volatile organic compounds; molc = mole of charge; H^+ = hydrogen ion; N = nitrogen; P = phosphorus; CTU_e = comparative toxic unit for ecosystems.

2.5.7 Life cycle interpretation

It is important to interpret LCA results by relating them to the goal and research question set up in the first phase of the study, and strictly limit interpretation scope according to the LCA approach being used (e.g. ALCA or CLCA). This is due to the fact that different research questions need to be addressed by using different LCA modelling approaches (Plevin et al. 2014). For example, ALCA is used to quantify environmental impacts and identify environmental hotspots, in order to find or research appropriate measures to tackle or mitigate environmental impacts to improve environmental performance of the systems. Interpretation of the LCA results derived from ALCA must be scoped within the system boundary defined (Plevin et al. 2014). In contrast, CLCA which is normally used to compare environmental performance resulting from a change in demand for a studied product, can produce comparative outputs between different systems (e.g. base system [i.e. without action] versus alternative systems [i.e. with action]).

2.6 Evaluation of different environmental impacts in LCA studies on dairy farming systems

This section provides an overview of the different environmental impacts conventionally assessed in LCA (Table 2.3), and discusses the extent to which each of these impacts is addressed in the published LCA studies of dairy farming systems listed in Tables A1 and A2 of the Supplementary material A.

2.6.1 Climate Change

General definition

Climate Change refers to changes in the pattern of the global climate resulting from an increase in the temperature of the atmosphere (Intergovernmental Panel on Climate Change 2007). This impact category considers GHG emissions arising from anthropogenic activities. In general, GWP for each of GHGs have been computed by the Intergovernmental Panel on Climate Change (2007) and used as the characterization factors to model the CC impact (EC-JRC-IES 2012). It is worth noting that the Intergovernmental Panel on Climate Change has recently revised the characterisation factors for each of GHGs (Myhre et al. 2013).

Key findings

All studies reviewed have assessed the impacts on CC (see Tables A1 and A2), although the name of this impact category varied (e.g. GWP and CF). In addition, all studies quantified this impact using old GWPs (characterization factors) derived from the Intergovernmental Panel on Climate Change (2007).

For the life cycle of dairy farming systems, three GHGs (methane [CH_4], nitrous oxide [N_2O] and CO_2) were the main contributors, collectively accounting for > 95% of the total impact (de Vries and de Boer 2010; Yan et al. 2011). The CC impacts per kg milk varied between studies, depending on a variety of factors, including functional units used, inventory models chosen, co-product handling methods deployed and dairy farming systems evaluated. The contribution of enteric CH_4 emissions was consistently the single largest component, whereas the shares of N_2O and CO_2 emissions differed, depending largely on farming practices and systems, e.g. confinement-based versus pasture-based (Yan et al. 2013a; O'Brien et al. 2014b).

As opposed to pasture-based systems which relied mainly on grazed grass, in confinement-based systems, the contribution of production of feed types other than pasture to the CC indicator results was generally substantial, driving relatively higher CC impacts compared with pasture-based systems (Jayasundara and Wagner-Riddle 2013; Kim et al. 2013). One of the main reasons for the relatively lower impact on CC for the pasture-based systems was that the CC impact of grazed grass was relatively lower than that for brought-in feed supplements (Mogensen et al. 2014).

In addition, LUC has been identified as a significant source of the CC impact in the life cycle of dairy farming systems, especially when deforestation was involved (Hutchings and Ledgard 2009; van Middelaar et al. 2013). Moreover, indirect GHG emissions (e.g. from a conversion of leached nitrate [NO_3^-] and volatilized ammonia [NH_3] to N_2O) significantly contributed to the CC impact (Ministries of Primary Industries 2013).

2.6.2 Ozone Depletion Potential

General definition

Ozone Depletion Potential refers to the potential impacts of ozone depletion substances (ODSs) released from anthropogenic activities on a reduction of the ozone layer in the stratosphere (World Meteorological Organization 1999). The reduction in stratospheric ozone layer potentially lead to increased incidence of skin cancers in human and damage of ecosystems (Norval et al. 2011). In general, chlorine atoms in chlorinated substances (e.g. tetrachloro-methane [CFC-10]) and bromide atoms in brominated substances (e.g. bromotrifluoro-methane [Halon 1301]) are effective in damaging ozone through irreversible reaction of these atoms with ozone molecules (Laube et al. 2014). It should be noted that production of chlorofluorocarbons (CFCs) has been phased out since the Montreal Protocol on substances that deplete the ozone layer came into force in 1989. This resulted in increased use of relatively lower potential ozone-damaging substances (e.g. hydrochlorofluorocarbons [HCFCs]) to replace CFCs. However, the HCFCs are also active to some extent in damaging the ozone layer.

Key findings

No dairy systems studies that were reviewed assessed this impact category. However, it has been reported that most ODSs in the life cycle of milk may in general be associated with manufacturing of pesticides and N fertilisers as well as the use (combustion) of fossil fuels (Arsenault et al. 2009; Castanheira et al. 2010).

2.6.3 Human Health Toxicity (non-cancer and cancer effects)

General definition

There are two impact categories associated with Human Health Toxicity: non-cancer effects and cancer effects, which are recommended to be characterized by the USEtox^{TM} models (Rosenbaum et al. 2008). The models transform the contributing substances into each of these two impact categories, taking into account their fate, exposures and potential effects. There is a long list of substances contributing to these two impact categories (Rosenbaum et al. 2008).

For most agricultural systems, it is recommended that pesticides and heavy metals should be accounted for when assessing environmental impacts because these two groups are the major contributors to Human Health Toxicity (Rosenbaum et al. 2008; Berthoud et al. 2011; Henderson et al. 2011). Seven heavy metals (cadmium [Cd], Chromium [Cr], Copper [Cu], Lead [Pb], Mercury [Hg], Nickel [Ni] and Zinc [Zn]) have been recommended to be inventoried when assessing environmental impacts of agricultural product systems (Nemecek and Kägi 2007). In general, these metals are contaminants in P fertilisers (ecoinvent Centre 2013; Agri-Footprint 2014). These heavy metals can reach ecosystems and inadvertently be consumed by human through the food web (Pizzol et al. 2011). In addition, Herrmann et al. (2013) reported that contributing substances associated with human health toxicity are substantially released from manufacturing of chemical fertilisers, implying that increased use of chemical fertilisers can exacerbate these impacts.

Dioxins and polychlorinated biphenyls (PCBs) are the most important groups of organic substances for the non-metal carcinogens, whereas Cd has the highest characterization factor for the metal carcinogens (Rosenbaum et al. 2008). Formaldehyde, an important constituent of pesticides, also plays a role in the cancer effects, implying that increased use of pesticides can exacerbate the cancer effects. In addition, Spinelli et al. (2013) reported that production of N fertilisers, especially ammonium nitrate, releases a number of substances contributing to the cancer effects, e.g. polycyclic aromatic hydrocarbon and PCBs.

Key findings

It should be noted that, at the time of the present study, studies associated with the impacts on non-cancer and cancer effects in the life cycle of dairy farming systems have not been published in scientific journals.

2.6.4 Particulate Matter

General description

Particulate Matter and Respiratory Inorganics (collectively called PM) refers to adverse respiratory effects on human health caused by inhalation of inorganic substances (Humbert et al. 2011). The PM contributing substances vary. However, based on the PM characterization model recommended by the European Food SCP Roundtable (2013), there are nine substances contributing to this impact category: NH_3, carbon monoxide (CO), nitric oxide (NO), nitrogen oxides (NO_x), particulate matter with diameter <10 μ, particulate matter with diameter < 2.5 μ, sulphur monoxide (SO), sulphur dioxide (SO_2) and sulphur trioxide (SO_3) (EC-JRC-IES 2012).

Key findings

For dairy farming systems, Place and Mitloehner (2010) reported that the impact on PM was partly associated with fine dusts resulting from animal locomotion and feed crop cultivation. In addition, Battini et al. (2014) reported that in the life cycle of a confinement-based dairy system, NH_3 was the single largest contributing substance for this impact category, and largely released from the on-farm stage.

2.6.5 Ionizing Radiation – human health effects

General definition

Ionizing Radiation - human health effects (IR) considers emissions of radioactive substances, and focuses on adverse effects on human health (EC-JRC-IES 2011). It has been reported that nuclear power plants, extraction of phosphate rocks and combustion of coals are the major contributors to release of IR substances (Frischknecht et al. 2000). This impact category assesses potential human health impacts of radioactive substances, taking into account their fate, exposures, potential effects and damages (EC-JRC-IES 2012).

Key findings

It should be noted that, at the time of the present study, this impact category has not been assessed in published studies on the life cycle of dairy farming systems.

2.6.6 Photochemical Ozone Formation Potential

General definition

Photochemical Ozone Formation Potential refers to the potential impacts associated with a presence of ozone and other reactive oxygen compounds in the troposphere as a result the oxidation of volatile organic compounds (VOCs) or CO in the presence of NO_x in sunlight (EC-JRC-IES 2011). The contributions of VOCs, CH_4 and NO_x to this impact category for the life cycle of agricultural products can be significant, even though emissions of contributing substances from the agricultural sector are much less than from the industrial sector (Laurent and Hauschild 2014).

Key findings

Place and Mitloehner (2010) reported that, in the USA, the main POFP contributing substances from dairy farming systems are VOCs and NO_x released largely from production of silage feeds. Likewise, in confinement-based systems, Battini et al. (2014) found that the major contributor to this impact indicator was NO_x emissions released mainly from manure storage, followed by the non-methane VOCs released from combustion of fossil fuels. In contrast, Arsenault et al. (2009) and Castanheira et al. (2010) reported that CH_4 was the single largest contributor to POFP, whereas Roer et al. (2013) reported that POFP contributing substances were released from production of conserved forage and supplementary feed in confinement-based systems. It should be noted that these studies have all made different assumptions, considered different system boundaries, used different inventory models and deployed different characterization models to estimate the impact on POFP.

2.6.7 Acidification Potential

General definition

Acidification Potential (AP) considers potentials of environmental impacts arising from contamination of natural ecosystems by acidifying substances released from anthropogenic activities (EC-JRC-IES 2011). As indicated in Table 2.3, the characterization model is based on the Accumulated Exceedance model according to Seppälä et al. (2006) and Posch et al. (2008). Based on this model, the contributing substances include NH_3, NO, NO_x, SO, SO_2 and SO_3 (EC-JRC-IES 2012) are computed and expressed in relation to the concentration (mole) of hydrogen ion (H^+).

Key findings

In general, there are three main acidifying substances released from dairy farming systems: NH_3, NO_x and SO_2, with NH_3 regarded as the single largest acidifying substance (Saggar et al. 2013). Different farming systems potentially influenced emissions of these acidifying gases. For example, pasture-based systems generally had lower emissions of NH_3 compared with confinement-based systems (Jarvis and Ledgard 2002). According to the studies reviewed, NH_3 was the single largest contributing substance (accounting for ~85% of the total impact), followed by SO_2 (accounting for ~10% of the total impact) and NO_x (accounting for ~5% of the total impact) (Guerci et al. 2013a; Battini et al. 2014).

2.6.8 Eutrophication Potential

There are three sub-impacts associated with Eutrophication Potential (EP), classified based on environmental compartments: Terrestrial Eutrophication Potential (TEP), Freshwater Eutrophication Potential (FEP) and MEP, as recommended by the European Food SCP Roundtable (2013). These impact categories are characterized using different characterization models and have different contributing substances.

Most reviewed studies used the characterization model of Guinée et al. (2002) without differentiating the EP impact according to affected environmental compartments (see Table A2). Based on these studies, the impact on EP in the life cycle of dairy farming systems was contributed by NH_3 emissions (accounting for ~44%) and NO_3^- leaching (accounting for ~44%) occurring mainly at the on-farm stage (Guerci et al. 2013a; Nguyen et al. 2013b).

2.6.8.1 Terrestrial Eutrophication Potential

General definition

Generally, TEP refers to the situation where terrestrial ecosystems are enriched by eutrophying substances that are released from anthropogenic activities (EC-JRC-IES 2011). This impact category is contributed by six N-based substances: NH_3, ammonium ion [NH_4^+], NO_3^-, NO, nitrite [NO_2^-] and NO_x (EC-JRC-IES 2012).

Key findings

It should be noted that, at the time of the present study, studies assessing the impact on TEP in the life cycle of dairy farming systems have not been published.

2.6.8.2 Freshwater Eutrophication Potential

General definition

In general, FEP refers to the enrichment of freshwater ecosystems by eutrophying substances released from anthropogenic activities (EC-JRC-IES 2011). In general, the growth of most aquatic plants in freshwater ecosystems is limited by P. Therefore, releases of three P-based substances are considered in this impact category: phosphate, phosphoric acid and P (EC-JRC-IES 2012).

Key findings

For the life cycle of dairy farming systems, Battini et al. (2014) reported that, in confinement-based systems, the two major contributing stages were: (i) the production of feeds (accounting for ~55%), and (ii) the on-farm stage (accounting for ~25%), caused mainly by P runoff. Similarly, Roer et al. (2013) reported that P-based emissions in confinement-based dairy systems occurred mainly at the stage associated with production of off-farm feeds.

2.6.8.3 Marine Eutrophication Potential

General definition

The situation where marine ecosystems are enriched by N-based substances is referred to as the impact on MEP. It has been documented that N-based substances are the major limiting substances contributing to proliferation of marine algal species (EC-JRC-IES 2011). This impact category is contributed by six N-based substances: NH_3, NH_4^+, NO_3^-, NO, NO_x and NO_2^- (EC-JRC-IES 2012).

Key findings

For the life cycle of dairy farming systems, Battini et al. (2014) reported that the major sources of N-based substances in confinement-based systems were N leaching and N runoff, resulting from the production of feed and from the on-farm stage, collectively accounting for ~80% of the total impact. Similarly, Roer et al. (2013) reported that, in confinement-based dairy systems, the contribution of brought-in feed production (conserved forage and supplementary feeds) played the biggest role in this impact category.

2.6.9 Ecotoxicity for Aquatic Freshwater

General definition

This impact category is characterized using the USEtox^{TM} model (Rosenbaum et al. 2008). The model accounts for the fate and effects of contributing substances on damage of freshwater ecosystems. The toxicity in this impact category refers to the toxic effect on aquatic freshwater species (e.g. algae, crustacean and fish) in water (EC-JRC-IES 2012). In general, for the life cycle of agricultural systems, pesticides and heavy metals are the major groups of the contributing substances (Berthoud et al. 2011; Yang and Suh 2014).

Key findings

Roer et al. (2013) assessed the impact on Ecotoxicity for Aquatic Freshwater in the life cycle of dairy farming systems in Norway, and reported that the main stage that released most contributing substances for this impact category was the production of brought-in feeds, especially feed grain (e.g. barley, oat and wheat) and soybean.

2.6.10 Resource Depletion

According to the recommendation of the European Food SCP Roundtable (2013), resource depletion should initially be focussed on two types of natural resources: (i) mineral and fossil, and (ii) freshwater.

2.6.10.1 Mineral and fossil

General definition

Generally, depletion of mineral and fossil materials is the focus of this impact category, calculated against the total quantities of mineral and fossil reserves (EC-JRC-IES 2011). For this impact category, a list of determining mineral and rare earth elements with different characterization factors has been identified. For the depletion of fossil resources, crude oil, hard coal, brown coal, natural gas and uranium are considered (EC-JRC-IES 2012).

Key findings

It should be noted that, at the time of the present study, studies focussing on the depletion of mineral and fossil resources in the life cycle of dairy farming systems using these characterization factors have not been published.

2.6.10.2 Freshwater

General definition

The depletion of freshwater is recommended to be characterized by taking into account a water scarcity index which is spatially dependent (Ridoutt and Pfister 2010), implying that characterization factors for the depletion of freshwater, in the area where there is limited availability of freshwater can be high (EC-JRC-IES 2012). The European Food SCP Roundtable (2013) recommended using characterization models and water scarcity index according to Ridoutt and Pfister (2010).

Key findings

Zonderland-Thomassen and Ledgard (2012) assessed the depletion of freshwater in the life cycle of dairy farming systems in New Zealand, using the characterization models according to Ridoutt and Pfister (2010), and reported that the depletion of freshwater (abstracted water) on a per kg FPCM was much lower in a non-irrigated dairy farm (0.165 L H_2O eq.) compared with that in an irrigated dairy farm (11.1 L H_2O eq.). In summary, the single largest contributor to the depletion of freshwater in dairy farming systems in New Zealand was the freshwater used for irrigation, accounting for ~94% of the total (Zonderland-Thomassen and Ledgard 2012). Ridoutt and Pfister (2010) reported that localized water footprint (the depletion of freshwater using local water stress index) of dairy systems in a major milk production region of Australia was 14.1 L per L of milk produced, and the major user was the on-farm stage which included the production of farm inputs, accounting for 83% of the total.

2.6.11 Land use impact

General definition

This impact category initially focuses on a change in soil organic matter (soil carbon) as a result of LU and LUC (Milà i Canals et al. 2007a). Generally, LU and LUC potentially leads to either benefits or problems associated with soil fertility and soil carbon dynamics. Note that this impact category is currently suitable for assessing agricultural and forest systems, where soil fertility plays a critical role (EC-JRC-IES 2012). The impact takes in to account the locations, timeframes and soil organic matter change after land occupation or land transformation.

Key findings

It should be noted that, at the time of the present study, there have been no published studies assessing this impact category in the life cycle of dairy farming systems.

2.7 Conclusions

The dairy sector is one of the most important sectors for the New Zealand economy. Over the past decade, the sector has increased its production capacity (milk production per hectare) by moving to more intensive systems. The intensification of New Zealand dairy farming systems has occurred through increased stocking rate coupled with increased use of off-farm inputs to support increased feed supply. Increased on-farm feed supply is usually achieved through increased use of N fertilisers and irrigation, along with improved pasture management skills, whereas increased used of brought-in feed supplements is an important source of off-farm feed supply. However, these intensification methods cause additional environmental emissions. Failing to account for these emissions and their environmental impacts increases risks associated with burden-shifting. Therefore, it is necessary to account for these emissions (including other upstream emissions) in order to ensure the environmental sustainability of dairy farming systems in New Zealand.

Currently, there are two types of LCA modelling: attributional and consequential. These two approaches address different questions and have different implications. Attributional modelling assesses environmental impacts for the life cycle of a product based on a status-quo. In contrast, environmental impacts arising from a change in demand for a product (or system changes) can be assessed using consequential modelling which accounts only for unit processes (marginal suppliers) that are actually affected or able to respond to such a change in demand. It is important to note that, as only marginal suppliers are accounted for in consequential modelling, its results do not represent total environmental impacts associated with a product system. This raises a question about whether there is a need for an alternative modelling approach that can generate information on the total (net) environmental impacts associated with prospective (future) product systems to support decision-making, where the focus is on envisioning alternative systems in the future.

Linked to the debate over attributional versus consequential modelling, choice of methods to handle co-products remains an issue in LCA studies. Choice associated with co-product handling methods can influence the LCA results and lead to incomparable and inconclusive LCA results from different studies. It is generally recommended to use system expansion/substitution to handle co-products when performing consequential modelling, and to use allocation methods when performing attributional modelling. For LCAs of dairy systems, the International Dairy Federation recommends to use allocation based on biophysical relationships in order to partition resource use and environmental emissions between the co-products of milk and dairy meat, and economic allocation for all inflows (e.g. feed). However, for system expansion/substitution, identification of appropriate displaced product systems can be challenging.

In summary, there are no comprehensive LCA studies of dairy farming systems. In addition, there is a lack of consensus on the applicability of different LCA modelling approaches to support different decision contexts. Therefore, research is needed on the comprehensive environmental LCA studies of dairy farming systems, and to elucidate the appropriateness of different LCA modelling approaches for the different decision situations related to dairy farming systems. Moreover, there is a lack of knowledge and understanding about the comprehensive environmental impacts associated with dairy farming intensification options to increase milk productivity, especially in pasture-based dairy systems. Given the current trends in New Zealand dairy farming systems towards greater intensification, these dairy farming systems are a good focus for investigation of these research issues.

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Supplementary material A

Table A1 A summary of published Life Cycle Assessment (LCA) studies (2012 – 2014) on a single impact category (Global Warming Potential and Carbon Footprint, collectively referred to the impact on Climate Change) of dairy farming systems using a cradle-to-farm gate perspective as a system boundary.

Reference	Country [a]	Goal	FU [c]	Dairy systems	Co-product handling methods		CC [e]
					Inflow	Outflow	
Dalgaard et al. (2014)	DK	Comparing different LCA approaches	1 kg ECM	Conventional	varied	varied	0.32 – 1.90
Flysjö et al. (2011b)	NZ	Investigating parameters contributing to milk CF [b]	1 kg ECM	NZ pasture-based	Economic	No allocation	1.00
	SW			SW concentrate-based	Economic	No allocation	1.16
Gollnow et al. (2014)	AU	Quantifying milk CF	1 kg FPCM	Conventional	Economic	Biophysical	1.11
Kristensen et al. (2011)	DK	Estimating GHG emissions associated with different dairy farming systems, and investigating effects of co-product handling methods	1 kg ECM	Conventional	NA [d]	Protein mass	0.99
					NA	Biophysical	0.91
					NA	Economic	1.06
					NA	Allocation based on a linear model	1.03
					NA	System expansion	0.94
				Organic	NA	Protein mass	1.02
					NA	Biological	0.90
					NA	Economic	1.10
					NA	Allocation based on a linear model	1.06
					NA	System expansion	0.96
Mc Geough et al. (2012)	CA	Determining GHG emissions associated with dairy farming system, and investigating effects of co-product handling methods	1 kg FPCM	Condiment-based	NA	No allocation	0.92
					NA	Economic	0.84
					NA	Biophysical	0.67
O'Brien et al. (2014a)	IR	Quantifying milk CF	1 kg FPCM	Grass-based	NA	Economic	1.11
O'Brien et al. (2014b)	IR	Comparing milk CF derived from different farming systems and investigating effects of co-product handling methods	1 kg FPCM	Grass-based	NA	System expansion	0.50
					NA	Biophysical	0.74
					NA	Protein mass	0.79
					NA	Economic	0.76
					NA	Mass	0.82
					NA	No allocation	0.84
	UK			Confinement-based	NA	System expansion	0.61
					NA	Biophysical	0.77
					NA	Protein mass	0.83
					NA	Economic	0.79
					NA	Mass	0.87
					NA	No allocation	0.88
	US			Confinement-based	NA	System expansion	0.64
					NA	Biophysical	0.79
					NA	Protein mass	0.84
					NA	Economic	0.84
					NA	Mass	0.88
					NA	No allocation	0.90
O'Brien et al. (2011)	IR	Comparing GHG emissions associated with different farming systems	1 kg raw milk	Grass-based	NA	Biophysical	0.81-0.90
Verge et al. (2013)	CA		1 L raw milk	Confinement-based	Mass	Biophysical	1.07
Yan et al. (2013a)	IR	Comparing milk CF derived from different farming systems	1 kg ECM	Pasture-based plus N fertiliser	Economic	Economic	1.04
				Pasture-based with high clover	Economic	Economic	0.87
Yan et al. (2013c)	IR	Quantifying milk CF	1 kg ECM	Grass-based	Economic	Economic	1.23

[a] AU: Australia, NZ: New Zealand, SW: Sweden, DK: Denmark, CA: Canada, IR: Ireland, UK: the United Kingdom and US: the United State of America; [b] CF: carbon Footprint; [c] FU: functional unit where ECM: energy corrected milk and FPCM: fat and protein corrected milk, [d] NA: data not available; [e] CC: Climate Change [kg CO_2 eq.].

Table A2 A summary of published Life Cycle Assessment (LCA) studies (2012 – 2014) on multiple impact categories of dairy farming systems using a cradle-to-farm gate perspective as a system boundary.

Reference	Country [a]	Goal	FU [b]	Dairy systems	Co-product handling methods		LCIA [d]	CC [e]	AP [f]	EP [g]
					Inflow	Outflow				
Bartl et al. (2011)	PE	Comparing environmental performances between the two dairy systems	1 kg ECM	Small holder, ryegrass (highland)	Mass	Economic	CML	13.78	14.13	15.47
			1 kg ECM	Small holder, confinement-based (lowland)				3.18	7.55	4.87
Battini et al. (2014)*	IT	Assessing environmental impacts of dairy system coupled with different mitigation options	1 kg FPCM	Confinement-based	NA [c]	No allocation	NA	1.21	-	-
					NA	Mass		1.18	-	-
					NA	Economic		1.13	-	-
Guerci et al. (2013a)	IT	Evaluating environmental impacts associated with intensive dairy systems	1 kg FPCM	Confinement - based	Economic	Economic	NA	1.30	19.7	9.01
Guerci et al. (2013b)	DK	Evaluating environmental impacts associated with different dairy farming systems	1 kg ECM	Organic and conventional	NA	Biophysical	NA	1.41	17.08	7.68
	DE		1 kg ECM	Pasture-based and confinement-based	NA	Biophysical	NA	0.94	12.75	6.15
	IT		1 kg ECM	Confinement-based	NA	Biophysical	NA	1.41	18.46	7.59
Iribarren et al. (2011)	ES	Assessing environmental impacts of dairy farming systems	1 kg raw milk	Confinement-based	Economic	Economic	ReCiPe	0.77	9.0	4.3
Meul et al. (2014)	BG	Estimating environmental impacts of milk	1 kg FPCM	Confinement	Economic	Economic	CML	1.04	13.57	3.78
Nguyen et al. (2013b)	FR	Evaluating effects of farming systems and co-product handling methods on environmental impacts of milk	1 kg FPCM	Grass only, Holstein-Friesian	NA	Biophysical	NA	1.33	11.26	4.37
					NA	Protein mass	NA	1.49	12.67	4.92
					NA	Economic	NA	1.55	13.14	5.34
					NA	System expansion	NA	0.97	10.39	2.92
				Confinement-based, Holstein-Friesian	NA	Biophysical	NA	1.17	9.85	4.71
					NA	Protein mass	NA	1.24	10.41	4.98
					NA	Economic	NA	1.27	10.69	5.11
					NA	System expansion	NA	0.87	8.67	3.72
O'Brien et al. (2012)	IR	Comparing environmental impacts between dairy farming system	1 kg FPCM	Grass-based	Economic	Biophysical	CML	0.87	6.9	3.4
			1 kg FPCM	Confinement-based	Economic	Biophysical		1.03	11.9	4.6
Roer et al. (2013)**	NO	Assessing environmental impacts of dairy systems	1 kg ECM	Confinement-based	NA	Economic	ReCiPe	1.53	-	-
Yan et al. (2013b)	IR	Assessing environmental impacts of dairy farming system	1 kg ECM	Grass-based	Economic	Economic	ReCiPe	1.34	14.4	-

[*] This study modelled a range of impact categories, but it used different characterization models and different units compared with the other studies; [**] this study modelled a range of environmental impacts and included the contribution of capital goods and veterinary medication, but it used different characterization models and different units compared with the other studies; [a] PE: Peru, IT: Italy, DK: Denmark, DE: Germany, ES: Spain, BG: Belgium, FR: France, NO: Norway and IR: Ireland; [b] FU: functional unit where ECM: energy corrected milk and FPCM: fat and protein corrected milk; [c] NA: data not available and [d] LCIA: Life Cycle Impact Assessment methods, CML (Guinée et al. 2002) and ReCiPe (Goedkoop et al. 2012); [e] CC: Climate Change [kg CO_2 eq.]; [f] AP: Acidification Potential [g SO_2 eq.]; [g] EP: Eutrophication Potential [g PO_4 eq.].

Chapter 3 Attributional Life Cycle Assessment of Pasture-Based Milk Production Systems in the Waikato Region, New Zealand

This chapter has been modified and accepted for publication in The International Journal of Life Cycle Assessment.

Abstract

Dairying is a relatively intensive livestock production system and contributes to a range of environmental impacts. In the southern hemisphere, dairy farming systems are based mainly on outdoor grazing of permanent pastures. The objectives of this study were: (i) to assess environmental profiles, and (ii) to identify environmental hotspots in a pasture-based dairy farming system. A cradle-to-farm gate Life Cycle Assessment of 53 dairy farms in the Waikato region, New Zealand was carried out, using one kg of fat and protein corrected milk as the functional unit. Twelve environmental impact categories were assessed: Climate Change (CC), Ozone Depletion Potential (ODP), Cancer effects (Cancer), Non-cancer effects (Non-cancer), Particulate Matter (PM), Ionizing Radiation (IR), Photochemical Ozone Formation Potential (POFP), Acidification Potential (AP), Terrestrial Eutrophication Potential (TEP), Freshwater Eutrophication Potential (FEP), Marine Eutrophication Potential (MEP) and Ecotoxicity for Aquatic Freshwater (Ecotox). Contribution and sensitivity analyses were performed to determine key hotspots and investigate potential changes in results due to methodological choices. The on-farm stage contributed >50% of the total result for 7 out of 12 indicators (CC, Non-cancer, PM, POFP, AP, TEP, MEP). The off-farm rearing of replacement animals contributed 11-20% to the total result for all indicators. The production of brought-in (from off-farm) feeds for use on the dairy farms contributed >10% to the indicator results for Non-cancer (25%), FEP (15%), MEP (12%) and Ecotox (19%). The manufacturing of agrichemicals for use on the dairy farms contributed >10% to the indicator results for ODP (26%), Cancer (26%), PM (19%), IR (46%), FEP (25%) and Ecotox (42%). The transportation of off-farm inputs for use on the dairy farms contributed >10% to the impacts on ODP (15%), IR (21%) and POFP (11%). Sensitivity analysis demonstrated the influence of choices associated with data sources, inventory and impact assessment models on the results. In conclusion, the off-farm activities together contributed >50% to 5 out of 12 impact indicator results, and >45% to a further two impact indicator results and 32% of the CC result. Therefore environmental improvement options should focus on both on-farm and off-farm activities. A focus on just one impact category (such as CC) risks ignoring the environmental hotspots that are revealed in a more comprehensive environmental assessment.

Keywords Environmental impact ● Life Cycle Assessment ● Milk ● New Zealand ● Pasture-based dairy farming

3.1 Introduction

Milk from dairy cows is an important livestock commodity that is increasingly produced and consumed worldwide. It has been anticipated that increases in world population and its wealth will lead to a doubling of demand for food (Smith et al. 2013), especially for animal-derived products (e.g. milk and meat) by 2050 (OECD/FAO 2013). Focusing on the dairy sector, the expansion of dairy farming has been identified as a significant contributor to environmental degradation (FAO 2006).

Life Cycle Assessment (LCA) is a system analytical approach that is increasingly used to quantify a range of environmental impacts, reveal environmental hotspots and identify environmental burden-shifting along the life cycle of a product (Finnveden et al. 2009). This approach has gained increasing attention for its role in informing environmentally-focused sustainable development, including in the dairy sector (FAO 2010). In recent years, LCA has been used to assess environmental impacts across dairy farming systems worldwide (de Vries and de Boer 2010). However, its application has been limited to a few environmental indicators, and there have been differences in methodological approaches.

Most LCA studies in dairying have been in the northern hemisphere and assessed confinement-based systems (de Vries and de Boer 2010), and few studies have examined dairy farms using pasture-based systems. Pasture-based dairy farming systems dominate in temperate southern hemisphere regions, including the largest dairy exporting country, New Zealand (OECD/FAO 2013). Also, globally, while a number of studies have examined greenhouse gas (GHG) emissions associated with dairy systems that are confinement-based (e.g. Thoma et al. 2013c; O'Brien et al. 2014a) and pasture-based (e.g. Flysjo et al. 2011b; Yan et al. 2013a), there have been only a few studies covering multiple environmental indicators for these systems (e.g. Thomassen et al. 2008b; Battini et al. 2014) and limited environmental indicators have been assessed for pasture-based systems (e.g. Basset-Mens et al. 2009b). Yet, from an LCA perspective, inclusion of a wide range of impact indicators is critically important in order to avoid burden-shifting (EC-JRC-IES 2010; European Food SCP Roundtable 2013).

Therefore, the objectives of the present study were to (i) assess environmental profiles and (ii) to identify environmental hotspots in a pasture-based dairy farming system, using New Zealand pasture-based dairy farming systems as a case study.

3.2 Methods

An attributional LCA approach was used. The methodology was compliant with the LCA framework (ISO 14040 and 14044) defined by the International Organisation for Standardization (International Organization for Standardization 2006a, 2006b). In addition, some methodological aspects specifically developed for LCA of dairy products and published in the International Dairy Federation (IDF) guidelines (International Dairy Federation 2010) were used, e.g. standardized functional unit and co-product handling methods. Inventory flows of the dairy farms studied were modelled using the SimaPro 8 software (Pré Consultants 2013).

3.2.1 Data source

The present study focused on a case study of pasture-based dairy farming systems in the Waikato region, New Zealand. The Waikato region is the largest dairy region in New Zealand, accounting for approximately 24% of total New Zealand milk production (DairyNZ 2013). Data were from 53 individual dairy farms in the Waikato region over one production year (2010/11), which had been collected into DairyBase (a web-based package that records dairy farm business information: http://www.dairynz.co.nz). The 2010/11 production year was typical for the Waikato climatic conditions, e.g. temperature, precipitation and rain distribution (DairyNZ 2011).

The herd size of the 53 dairy farms ranged from 128 to 1,135 milking cows per herd with an average of 448 cows per herd, and an effective dairy farmland area of 50 to 427 ha with an average of 165 ha per herd. Holstein-Friesian and Jersey × Holstein-Friesian crossbred cows predominated.

Data associated with other farm attributes are presented in Table 3.1. The mean and range of milk production from the dairy farms studied were marginally higher than the average of dairy farming systems in the Waikato region recorded in the same production year, while the stocking rates for the two data sources were similar (see Table 3.1). It was assumed that all farms had a constant calving rate of 80%, an average calving rate of dairy farms in the Waikato region, New Zealand (StatisticsNZ 2013).

Table 3.1 Means, standard deviations (SD), minimum (Min) and maximum (Max) values of farm attributes of the 53 studied dairy farms in the Waikato region, New Zealand for the year 2010/11; a per-ha basis refers to the on-farm stage only.

Farm attributes	Unit	Mean	SD	Min	Max	Ave [e]
(1) Stocking rate	LU [a]/ha	2.73	0.55	1.88	4.02	2.72
(2) Replacement rate	%	23	5	16	35	NA [f]
(3) Fertilisers						
• Nitrogen (N)	kg N/ha	142	85	0	255	NA
• Phosphorus (P)	kg P/ha	49	36	0	168	NA
• Potassium (K)	kg K/ha	45	32	0	110	NA
(4) Total dry matter intake	kg/ha	11,300	2,447	7,786	16,726	NA
• Estimated pasture intake	kg/ha	9,038	1,545	6,596	12,640	NA
• Total brought-in feed offered	kg/ha	2,744	1,907	0	6,192	NA
o Palm kernel expeller	kg/ha	1,643	1,513	0	4,783	NA
o Maize silage	kg/ha	893	867	0	2,857	NA
o Concentrate feed	kg/ha	34	178	0	924	NA
o Pasture silage	kg/ha	160	234	0	816	NA
o Pasture hay	kg/ha	13	42	0	164	NA
(5) % brought-in feed [b]	%	21	13	0	41	NA
(6) FCE [c]	-	1.13	0.08	0.96	1.29	NA
(7) FPCM [d] production	kg/cow	4,730	838	3,190	6,139	4,479
	kg/ha	12,851	3,227	8,024	20,114	12,362

[a] LU: livestock unit (1 LU = 1 dairy cow at 500 kg body weight); [b] (% brought-in feed = [total brought-in feed intake/total feed intake] × 100); [c] feed conversion efficiency (FCE) was computed as total milk production (kg) per ha/total feed intake of milking cows (kg dry matter) per ha; [d] FPCM: fat and protein corrected milk; [e] average physical traits of dairy farms in the Waikato region, based on 3,566 farms for the year 2010/11 (DairyNZ 2011); [f] not available.

3.2.2 Functional unit and system boundary

The functional unit was one kg of fat and protein corrected milk (FPCM), computed using the equation in the IDF guidelines (International Dairy Federation 2010): FPCM (kg) = milk yield (kg) × [(0.1226 × fat%) + (0.0776 × protein%) + 0.2534]. A cradle-to-farm gate perspective was used as a system boundary (see Figure 3.1). The products generated were milk and meat derived from culled cows and surplus calves slaughtered at about one week after calving (Flysjö et al. 2011a).

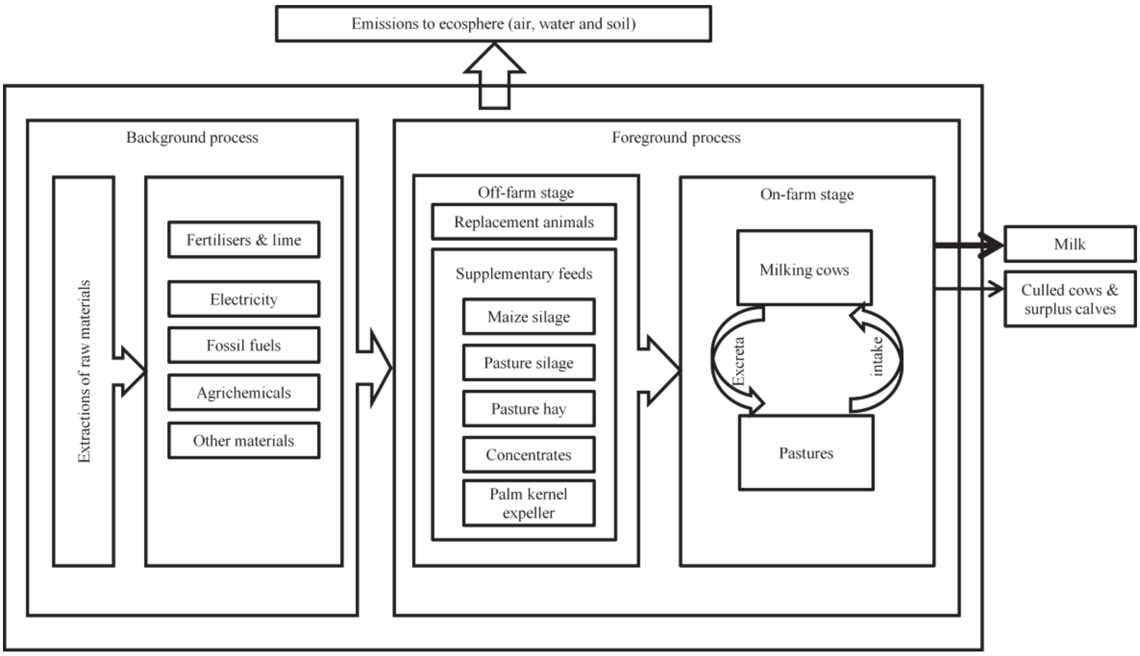

Figure 3.1 System boundary and simplified material flows in dairy farming systems studied.

3.2.3 Co-product handling methods

In general, milk and meat are the only two co-products generated from New Zealand dairy farming systems. All animal excreta remains on-farm (approximately 95% returned directly through grazing and 5% collected during milking and applied back onto pasture; Ledgard and Brier (2004)). Therefore no manure is exported as a co-product.

Allocation between dairy co-products has been widely discussed in the literature (e.g. Nguyen et al. 2013b; Kiefer et al. 2015). for this study, following the IDF guidelines (International Dairy Federation 2010), a physiological relationship was used to allocate environmental burdens between milk and meat. This allocation method reflects energy demands to produce milk and meat by dairy animals (Thoma et al. 2013a). Economic allocation was used to handle co-products associated with the production of brought-in supplementary feeds, as recommended by the IDF guidelines (International Dairy Federation 2010).

3.2.4 Life cycle inventory methods

3.2.4.1 General inventory method

The environmental impacts for each of the 53 dairy farms were modelled individually. The processes involved in the cradle-to-farm-gate life cycle of the milk were divided into foreground and background processes (see Figure 3.1). The foreground processes were divided into off-farm and on-farm stages. The off-farm stage and background processes were categorized as: (i) rearing of replacement animals (i.e. from weaning to near-calving, as it is normal practice in the Waikato to rear replacement animals off-farm), (ii) production of brought-in feeds for use on dairy farms, (iii) manufacturing of agrichemicals for use on dairy farms, and (iv) transportation of off-farm inputs for use on dairy farms. It should be noted that emissions associated with manufacturing of agrichemicals (e.g. fertilisers and pesticides), production of brought-in concentrate feeds and transportation of all off-farm inputs used for both rearing replacement animals and/or production of brought-in feeds were included in these two categories. Additionally, electricity generation and production of fossil fuels (diesel and petrol) for energy sources used on-farm and off-farm have been accounted for and were incorporated into the relevant on-farm or off-farm stage. Emissions associated with the background processes were derived from the ecoinvent v3 database (ecoinvent Centre 2013).

Emissions associated with the rearing of replacement heifers were modelled according to Basset-Mens et al. (2009b), except that recent information on feed requirements for rearing heifers (DairyNZ 2014b) was used. Total amounts of farm inputs and outputs, inventory models and estimated environmental impacts associated with the rearing of heifers and the production of brought-in feeds are presented in Tables B1, B2 and B3 of the Supplementary material B, respectively.

Computation of N in animal excreta was based on total N intake less total N output in milk and N retained in animals. Chemical composition and digestibility of the feeds were from Kolver (2000). The gross energy contents of individual feeds, which were required for estimation of enteric methane (CH_4) emissions, were calculated based on protein, fat and carbohydrate contents, as recommended by Waghorn (2007). Data associated with chemical composition of the feeds are presented in Table B4 of the Supplementary material B.

Methodologies used to estimate substances released from animal and manure management were based on the guidelines provided by the New Zealand Ministry for Primary Industries (2013) (see Table B2 for more details). In the case where emissions of some relevant processes and products were not available specifically for New Zealand dairy farming systems, such emissions were derived from the ecoinvent v3 database, especially for transportation and production of chemical fertilisers and pesticides. Proxies of such processes/products are presented in Table B5 of the Supplementary material B.

The OVERSEER® nutrient budgeting model (Wheeler et al. (2003) was used to estimate nitrate (NO_3) leaching and phosphorus (P) losses to water from all land uses (i.e. crop production, rearing of dairy replacements and on dairy farm). This model takes into account nutrient inputs and outputs, as well as the influence of relevant site-specific factors, e.g. precipitation, topography and soil properties (Wheeler et al. 2003).

Inputs, outputs and emissions associated with production of brought-in feeds were modelled, including for palm kernel expeller (PKE), maize silage, concentrate feeds, pasture silage and pasture hay. All brought-in feeds were assumed to be produced off-farm. A mixture of 60% maize grain and 40% barley grain was used as a proxy for concentrate feed since they were the major components of the concentrate feed for dairy cattle in New Zealand (Wales et al. 2009). Data associated with the cultivation methods and resources used for production of these brought-in feeds were derived from industry sources (S. F. Ledgard, unpublished) (see Table B1), except for PKE which was obtained from the Agric-Footprint database (Agri-Footprint 2014).

Ammonia (NH_3) emissions from N fertilisers for crop production were quantified based on Nemecek et al. (2014), except for estimation of New Zealand specific crop-residues which was based on Thomas et al. (2011). Other inventory models are given in Table B2.

Emissions associated with substances released from production of ensilage feeds (i.e. maize silage and pasture silage) were estimated using emission factors recommended by Del Prado et al. (2011) (for NH_3 and nitrogen oxides (NO_x)) and Rotz et al. (2013) (for volatile organic compounds (VOCs)). It is important to note that silage losses in the form of seepage can be a significant source of environmental impacts from silage stacks (Gebrehanna et al. 2014). However, as the fate of these losses has not yet been studied and the proportion of silage conserved in stacks was uncertain, these emissions were not accounted for in the present study.

Note that environmental emissions associated with land use change and soil carbon change were not accounted for in the present study, except for PKE where direct land use change were accounted for.

3.2.4.2 Inventory for release from pesticides

Pesticides used for the production of brought-in feeds (maize silage, maize grain and barley grain) and for the operation of dairy farming were derived from Manktelow et al. (2005). The emissions associated with pesticide manufacturing were derived from the ecoinvent v3 database.

Release of pesticide active ingredients into different environmental compartments (i.e. air, water and soil) was quantified using the following methods. The pesticide active ingredients released into the air were estimated based on the vapour pressures of individual active ingredients (see Table B6 of the Supplementary B for more details), as recommended by the European Environment Agency 'Air Pollutant Emission Inventory Guidebook' (Webb et al. 2013). Pesticide released into water was based on Kellogg et al. (2002) at 5% of total pesticides applied. The remaining pesticides were assumed to be deposited onto agricultural soils, and capped at 85%, as recommended by Audsley et al. (2003).

3.2.4.3 Inventory for release of heavy metals

Heavy metals can contribute to toxicity impact categories (Rosenbaum et al. 2008). In agricultural systems, Nemecek and Schnetzer (2011) recommended to include seven heavy metals; i.e. cadmium (Cd), chromium (Cr), copper (Cu), lead (Pb), mercury (Hg), nickel (Ni) and zinc (Zn). Therefore, data were collected for these heavy metals. In general, the major sources of these heavy metals were chemical fertilisers, especially P fertilisers.

The concentrations of heavy metals in chemical fertilisers vary according to the raw materials used in manufacturing processes. In this case study, the heavy metal contents in fertilisers were derived from a major fertiliser company in New Zealand (S.F. Ledgard, unpublished).

For inventory analysis associated with heavy metals, as there are no widely accepted models to estimate releases of heavy metals to environmental compartments, it was assumed that all heavy metals released at the on-farm stage were deposited onto agricultural soils. This assumption was also applied for the production of feed crops and pastures for the rearing of heifers, and has been used in a number of recent LCA studies (e.g. Yang 2013; Astudillo et al. 2014; Niero et al. 2014). However, it is important to note that this assumption can potentially underestimate toxicity since releases of heavy metals to water generally have higher impacts than those released to agricultural soils (Rosenbaum et al. 2008).

3.2.4.4 Inventory for emissions associated with fossil fuels and electricity

Data on the use of fossil fuels (diesel and petrol) and electricity for the individual dairy farms studied were not available, therefore national average data from StatisticsNZ (2014) were used as proxies (see more details in Table B7 of the Supplemtary material B). To estimate emissions associated with the electricity grid mix in New Zealand, electricity generation processes derived from the ecoinvent v3 database (without any modification) were used as proxies. These processes were combined according to the proportion of electricity sources for the year 2011 from the New Zealand Ministry of Business Innovation and Employment (MBIE 2013), comprising 58% from hydropower, 18% from natural gas, 13% from geothermal, 5% from coal, 4% from wind power and 1% from bioenergy.

3.2.5 Impact assessment methods

The impact categories recommended by the European Food Sustainable Consumption and Production Round Table (European Food SCP Roundtable 2013) were used (see Table 3.2). However, due to a lack of relevant inventory data, the impacts associated with land use and depletion of freshwater, mineral and fossil resources were not included in the present study. For GHG emissions, the recent characterization factors for GHGs over a 100-year time horizon were used, as recommended by the Intergovernmental Panel on Climate Change (Myhre et al. 2013).

Table 3.2 Impact categories used in the present study, based on the European Food SCP Roundtable (2013). The impacts superscripted by * were not included in the present study.

Impact category	Abbreviation
Climate Change	CC
Ozone Depletion Potential	ODP
Human Health Toxicity, cancer effects	Cancer
Human Health Toxicity, non-cancer effects	Non-cancer
Particulate Matter	PM
Ionizing Radiation – human health effects	IR
Photochemical Ozone Formation Potential	POFP
Acidification Potential	AP
Eutrophication – terrestrial ecosystems	TEP
Eutrophication – freshwater ecosystems	FEP
Eutrophication – marine ecosystems	MEP
Ecotoxicity for Aquatic Freshwater	Ecotox
Resource Depletion-mineral, fossil*	RD (mineral, fossil)
Resource Depletion-water use*	RD (water use)
Land Use Impact*	LU

3.3 Results and discussion

The allocation factor for partitioning environmental burdens between milk and meat was calculated for each farm separately. The mean allocation factor for milk was 82%, with the upper and lower 95% confidence intervals of 81% and 83%, respectively, and the remainder was allocated to meat.

3.3.1 Environmental profiles

The means, standard deviations and 95% confidence intervals of the environmental indicators (per kg FPCM) are presented in Table 3.3. It can be seen that there is variation in results within the impact categories, and this implies that there may be potential for improvement in at least some pasture-based dairy farms in order to reduce environmental impacts.

Table 3.3 Means, standard deviations (SD) and 95% confidence intervals (95% CI) of environmental impacts (per kg FPCM) of 53 dairy farms in the Waikato region, New Zealand for the year 2010/11.

Impact category [a]	Units [b]	Mean	SD	95% CI	
				Lower limit	Upper limit
CC	kg CO_2 eq.	0.80E+00	0.08E+00	0.78E+00	0.82E+00
ODP	kg CFC-11 eq.	1.02E-08	0.17E-08	0.97E-08	1.06E-08
Cancer	CTU_h	1.00E-08	0.40E-08	0.89E-08	1.11E-08
Non-cancer	CTU_h	2.60E-07	0.95E-07	2.33E-07	2.86E-07
PM	kg $PM_{2.5}$ eq.	4.60E-04	0.74E-04	4.40E-04	4.80E-04
IR	kg U^{235} eq.	1.06E-02	0.29E-02	0.98E-02	1.14E-02
POFP	kg NMVOC eq.	2.58E-03	0.28E-03	2.50E-03	2.66E-03
AP	molc H^+ eq.	1.53E-02	0.19E-02	1.48E-02	1.59E-02
TEP	molc N eq.	6.55E-02	0.80E-02	6.33E-02	6.77E-02
FEP	kg P eq.	0.96E-04	0.19E-04	0.91E-04	1.02E-04
MEP	kg N eq.	2.67E-03	0.43E-03	2.55E-03	2.79E-03
Ecotox	CTU_e	1.23E+00	0.34E+00	1.14E+00	1.32E+00

[a] Abbreviations for impact categories refer to as in Table 3.2; [b] CO_2 = Carbon dioxide, eq. = equivalent, CFC-11 = Trichlorofluoro-methane, CTU_h = Comparative toxic unit for humans, $PM_{2.5}$ = Particulate matter less than 2.5 μ in diameter, kBq = kilobecquerel, U^{235} = Uranium-235, NMVOC = Non-methane volatile organic compounds, molc = mole of charge, H^+ = Hydrogen ion, N = Nitrogen, P = Phosphorus and CTU_e = Comparative toxic unit for ecosystems.

The CC indicator results in the present study (0.78-0.82 kg CO_2 eq. per kg FPCM) are within the ranges of those found in other LCA studies. For example, these have been calculated as 0.65-0.93 kg CO_2 eq. per kg FPCM in New Zealand pasture-based dairy systems (Basset-Mens et al. 2009b) and, at the lower end, 0.87-1.72 kg CO_2 eq. per kg FPCM in Irish pasture-based dairy systems (O'Brien et al. 2014a) and 0.80-1.50 kg CO_2 eq. per kg FPCM in Italian confinement-based dairy systems (Fantin et al. 2012; Battini et al. 2014). For other impact categories, most indicator results in the present study are also similar to those found in Italian confinement-based dairy systems, as reported by Battini et al. (2014), including POFP (2.58E-03 vs 2.64E-03kg NMVOC eq. per kg FPCM, respectively), AP (1.53E-02 vs 1.23E-02 molc H^+ eq. per kg FPCM, respectively) and FEP (0.96E-04 vs 1.15E-04 kg P eq. per kg FPCM, respectively). The PM results were higher (4.60E-04 vs 3.88E-05 kg $PM_{2.5}$ eq. per kg FPCM, respectively) and the MEP results were slightly lower (2.67E-03 vs 8.75E-03 kg N eq. per kg FPCM, respectively). Apart from differences in farming systems, it should be noted that the differences in inventory procedures, allocation methods and characterization models can partly explain differences in these results.

3.3.2 Environmental hotspots

As shown in Figure 3.2, the contribution of the different stages in the cradle-to-farm gate life cycle of milk for the range of environmental impact categories. The on-farm stage was dominant (contributing >50% of the total impacts) for the CC, Non-cancer, PM, POFP, AP, TEP and MEP impact categories, and contributed 30-50% of the ODP, Cancer and FEP impact category results; it contributed less than 30% to the IR and Ecotox impact category results.

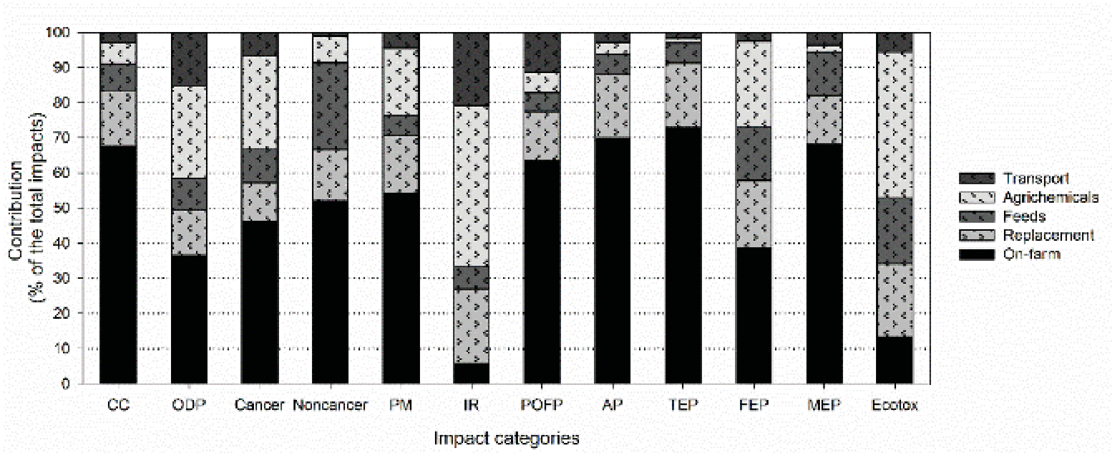

Figure 3.2 The contribution analysis of the cradle-to-farm gate life cycle of New Zealand milk for the environmental impacts studied, representing the mean for 53 farms in the Waikato region, New Zealand. Stages were divided into (i) on-farm emissions (On-farm), (ii) rearing of replacement animals (Replacement), (iii) production of brought-in feeds (Feeds), (iv) manufacturing of chemicals (Agrichemicals) and (v) transportation of off-farm inputs, e.g. brought-in feeds, chemical fertilisers and fossil fuels (Transport). Abbreviations for impact categories refer to as in Table 3.2.

The rearing of heifers and the production of brought-in feeds for use on a dairy farm contributed on average approximately 16% (range 11-20%) and 11% (range 5-25%) respectively, of the total results for all impact categories. In addition, the production of brought-in feeds was important for the Non-cancer (25% of the total result), FEP (15% of the total result) and Ecotox (19% of the total result) impact categories.

The manufacturing of agrichemicals for use on a dairy farm played a role in the impacts on ODP (26% of the total result), Cancer (26% of the total result), PM (19% of the total result), IR (46% of the total result), FEP (25% of the total result) and Ecotox (42% of the total result).

The contribution of transport of inputs for use on dairy farms to all impact categories was small (2-7% of the total impacts), except for the impacts on ODP (15% of the total impacts), IR (21% of the total impacts) and POFP (11% of the total impacts).

The environmental hotspots for each impact category in the cradle-to-farm gate life cycle of New Zealand milk for the average of 53 farms are discussed in the following sections.

3.3.2.1 Climate Change

The on-farm stage contributed 68% of the total impact result, followed by the rearing of heifers (16% of the total impacts), the production of brought-in feeds (8% of the total impacts), manufacturing of agrichemicals (6% of the total impacts) and transportation of farm inputs (3% of the total impacts). Methane, nitrous oxide (N_2O) and carbon dioxide (CO_2) accounted for 59%, 20% and 21% of the total impacts, respectively. It is worth noting that CH_4 was mainly from enteric fermentation, N_2O was mainly from N in animal excreta and N fertilisers, and CO_2 was mainly associated with the use of fossil diesel.

This midpoint indicator has been extensively assessed worldwide. Almost all of dairy impact assessment studies have assessed this indicator and reported that three main GHGs (i.e. CH_4, N_2O and CO_2) contribute to this indicator. Of these GHGs, CH_4 is usually the biggest contributor, originating mainly from enteric fermentation (de Vries and de Boer 2010; Battini et al. 2014).

The proportions of N_2O and CO_2 can vary, depending on farming practices. For example, dairy farms that use large amounts of brought-in feeds usually have a higher proportion of CO_2 (O'Brien et al. 2014b). This is due to the fact that modern feed-crop production practices are mechanized and require relatively large amounts of fossil fuels (Safa et al. 2011; Van linden and Herman 2014). In addition, increased use of N fertilisers can cause increased CO_2 since its production processes require large amounts of fossil energy and release considerable amounts of GHGs (Ledgard et al. 2011; Hasler et al. 2015).

3.3.2.2 Ozone Depletion Potential

The contributions of the on-farm stage and manufacturing of agrichemicals dominated (37% and 26% of the total impacts, respectively), followed by transportation of farm inputs (15% of the total impacts) and the rearing of replacement animals (13% of the total impacts). The production of brought-in feeds was a minor contributor to this impact category, accounting for 9% of the total impacts.

Bromotrifluoro-methane (Halon 1301), bromochlorodifluoro-methane (Halon 1211) and tetrachloro-methane (CFC-10) were the three main ozone depleting substances, accounting for 66%, 15% and 7% of the total impacts, respectively. Halon 1301 originated mainly from the manufacturing of fossil energy (i.e. diesel and petrol).

Manufacturing of chemical fertilisers also was responsible for the emissions of Halon 1301 and Halon 1211. Manufacturing of pesticides was responsible for CFC-10 emissions, and generation of electricity was associated with the emissions of CFC-10 and Halon 1211.

There are no dairy-related LCA studies assessing the impact on ODP. In the present study, ozone depleting substances (e.g. Halon and CFC) were released mainly from manufacturing of agrichemicals (particularly chemical fertilisers and pesticides).

3.3.2.3 Human Health Toxicity-cancer effects

The on-farm stage was the largest contributor to this impact category (46% of the total impacts), followed by manufacturing of agrichemicals (26% of the total impacts). The rearing of replacement animals and the production of brought-in feeds each accounted for approximately 10% of the total impacts. Transportation of off-farm inputs for use on a dairy farm contributed approximately 7% of the total impacts to this impact category.

Chromium was the largest contributor (94% of the total impacts) for this impact category, whilst Hg and Cd accounted for 3% and 2% of the total impact, respectively. It should be noted that Cr (68% of total impacts) was associated with the use of chemical fertilisers at the on-farm stage. The manufacturing and use of chemical fertilisers, especially P fertilisers, was the largest single contributing source overall.

Note that the characterization models (USEtox^{TM} v1.01) regarding organic toxicants provide greater certainty of results, while the models for inorganic toxicants remain highly uncertain and therefore, it has been recommended that the corresponding results (especially when including the impacts of inorganic toxicants) must be interpreted with care (Rosenbaum et al. 2008; Pizzol et al. 2011).

3.3.2.4 Human Health Toxicity-non-cancer effects

This indicator was dominated by the on-farm stage (52% of the total impacts), followed by the production of brought-in feeds (25% of the total impacts) and the rearing of replacement animals (15% of the total impacts). The manufacturing of agrichemicals (8% of the total impacts) and transportation of farm inputs (1% of the total impacts) were negligible.

Zinc (59% of the total impacts), Cd (20% of the total impacts), Hg (14% of the total impacts) and Pb (2% of the total impacts) were the main substances that contributed to this impact category. Approximately 50% of the total Zn emissions and 87% of the total Cd emissions originated from the use of chemical fertilisers at the on-farm stage.

3.3.2.5 Particulate Matter

The on-farm stage contributed most to this impact category (54% of the total impacts), followed by manufacturing of agrichemicals (19% of the total impacts) and the rearing of replacement animals (16% of the total impacts). There were minor contributions from the production of brought-in feeds (6% of the total impacts) and transportation of farm inputs (4% of the total impacts).

Ammonia, fine particulates (diameter <2.5 μ) and sulphur dioxide (SO_2) were the main substances that contributed to this impact category, accounting for 67%, 23% and 9% of the total impacts, respectively.

It is important to note that NH_3 emissions originated mainly from excreta and N fertilisers from the on-farm stage (75% of total NH_3 emissions) and the rearing of replacement animals (19% of total NH_3 emissions). The fine particulates and SO_2 were released mainly from the manufacturing of chemical fertilisers.

Place and Mitloehner (2010) reported that the impact of dairy farming systems on the PM indicator (e.g. fine dust) was partly associated with animal locomotion and feed production. However, in the present study, dusts originating from movements of animals and tillage activities were not accounted for due to unavailability of data.

Battini et al. (2014) reported that NH_3 was the biggest contributing PM substance in confinement systems in Italy. While confinement systems are known to generate far higher NH_3 emissions than grazing systems, e.g. Jarvis and Ledgard (2002), the latter can still be significant. In the present study, approximately three-quarters of NH_3 emissions originated from urinary and fertiliser N at the on-farm stage, while the rearing of heifers contributed near a quarter of the total NH_3 emissions.

3.3.2.6 Ionizing Radiation-human health effects

Manufacturing of agrichemicals was the largest contributor to this impact category (46% of the total impacts). The rearing of replacement animals and transportation of farm inputs had equal impacts on this impact category (21% of the total impacts). The production of brought-in feeds was a minor source, accounting for 6% of the total impacts.

Radioactive substances, including Radon^{222} and Carbon^{14} were the two main contributors, accounting for 54% and 43% of the total impacts, respectively. The release of these two substances was associated mainly with the manufacturing of P fertilisers that were assumed to be imported from Europe (due to insufficient NZ-specific data); in Europe, the production system was associated with the use of electricity that was partially generated by nuclear power plants.

3.3.2.7 Photochemical Ozone Formation Potential

The on-farm stage played a major role in this impact category (64% of the total impacts). The contributions of the rearing of replacement animals (14% of the total impacts), transportation of farm inputs (11% of the total impacts), manufacturing of agrichemicals (6% of the total impacts) and the production of brought-in feeds (5% of the total impacts) were small.

This impact category was influenced predominantly by VOCs and NO_x, accounting for 56% and 30% of the total impacts, respectively. CH_4 was responsible for about 7% of the total impacts.

The release of VOCs was mainly from manure of dairy cows at the on-farm stage (47% of the total impacts) and manure from replacement animals (8% of the total impacts). Nitrogen oxides originated mainly from the use of fossil diesel, while enteric fermentation released CH_4.

It has been reported that large numbers of non-methane volatile organic compounds contribute to this environmental indicator (Laurent and Hauschild 2014). Place and Mitloehner (2010) reported that POFP contributing substances (e.g. VOCs and NO_x) in dairy farming systems in the USA (predominantly confinement systems) were associated largely with production of silage feeds. This was due to the fact that dairy farming systems in the USA depend substantially on ensiled feeds, especially maize silage. Likewise, in a confinement system in Italy, Battini et al. (2014) found that the major contributor to this indicator was NO_x emissions released mainly from manure storage, followed by non-methane volatile organic compounds released from combustion of fossil fuels. It should be noted that VOCs emissions derived from silage making were not accounted for in the latter study.

Similarly, in the present study, the main contributing substances were VOCs and NO_x. However, the main hotspots were associated with animal excreta for VOCs, and combustion of fossil fuels for NO_x emissions. This is because less ensilaged feed is used in New Zealand pasture-based dairy systems, and so the hotspots for this indicator in this case study were the on-farm excreta and the use of fossil fuels.

3.3.2.8 Acidification Potential

The contribution of the on-farm stage was substantial, accounting for 70% of the total impacts, followed by the rearing of replacement animals (18% of the total impacts). The production of brought-in feeds, manufacturing of agrichemicals and transportation contributed 6%, 3% and 3% of the total impacts, respectively.

Ammonia was the dominant contributor to this impact category (91% of the total impacts). The other two acidifying substances (i.e. SO_2 and NO_x) played a minor role, accounting for 5% and 4% of the total impacts, respectively.

Nitrogen in excreta and fertilisers at the on-farm stage and the rearing of replacement animals contributed 75% and 19% of the total NH_3 emissions, respectively. Sulphur dioxide was released from manufacturing of the chemical fertilisers, while NO_x originated from combustion of fossil fuels.

In the present study, the largest single substance contributing to this indicator was NH_3 which was released mainly from the on-farm stage. Even though there are differences regarding farming systems, e.g. higher NH_3 loss from confinement systems (Rotz et al. 2009), inventory methods and characterization models, most dairy LCA studies (e.g. Thomassen et al. 2008b; Guerci et al. 2013a; Battini et al. 2014) have indicated that NH_3 was the dominant hotspot acidifying substance.

3.3.2.9 Terrestrial Eutrophication Potential

Similar to the impact on AP, this impact category was dominated by the on-farm stage (73% of the total impacts), followed by the rearing of replacement animals (18% of the total impacts). The contributions of the production of brought-in feeds (6% of the total impacts), manufacturing of agrichemicals (1% of the total impacts) and transportation of farm inputs (2% of the total impacts) were small.

The two main eutrophying substances (NH_3 and NO_x) accounted for 95% and 5% of the total impacts, respectively. Major sources of NH_3 and NO_x were as described for the impact on AP.

3.3.2.10 Freshwater Eutrophication Potential

The contributions of the on-farm stage along with manufacturing and use of P fertilisers to this impact category were substantial, accounting for 39% and 25% of the total impacts, respectively. The rearing of replacement animals (19% of the total impacts) and the production of brought-in feeds (15% of the total impacts) were also significant. Transportation of farm inputs contributed only 2% of the total impacts.

The release of P compounds was responsible for 100% of the total impacts for this impact category.

Phosphorus losses from soils (via runoff and leaching) at the on-farm stage and the rearing of replacement animals were the major sources for this impact category, accounting for 37% and 11% of the total impacts, respectively. In addition, the release of P compounds during the manufacturing of P fertilisers was responsible for 24% of the total impacts (data derived from ecoinvent v3 database). The two major sources of P loss from land were from application of P fertilisers and manures. While the former is of greater importance in grazed systems, the latter is significant in confinement systems (Rotz et al. 2009).

3.3.2.11 Marine Eutrophication Potential

This impact category was dominated largely by the on-farm stage (68% of the total impacts), followed by the rearing of replacement animals (14% of the total impacts) and the production of brought-in feeds (12% of the total impacts). The contributions of transportation of farm inputs (4% of the total impacts) and manufacturing of agrichemicals (2% of the total impacts) were small.

This impact category was contributed solely by nitrogenous substances. The release of NO_3 to water (72% of the total impacts), and NH_3 (17% of the total impacts) and NO_x (12% of the total impacts) to the atmosphere were the three main contributors. It should be noted that NO_3 leaching at the on-farm stage, which was mainly from animal urine deposition, accounted for 53% of the total impacts. Again, the emissions of NH_3

originated mainly from excretal N and volatilization of N fertilisers, while NO_x was emitted from the use of fossil diesel.

In a confinement system, Battini et al. (2014) reported that the main substance contributing to the impact on MEP was NO_3 which was associated with the on-farm stage. All-year grazing systems can result in relatively higher NO_3 leaching than well-managed confinement systems (Rotz et al. 2009) and therefore, N management is particularly important in grazing systems (Ledgard et al. 2009).

3.3.2.12 Ecotoxicity for Aquatic Freshwater

The largest contributor to this category was manufacturing of agrichemicals (42% of the total impacts), followed by the rearing of replacement animals (21% of the total impacts), the production of brought-in feeds (19% of the total impacts) and the on-farm stage (13% of the total impacts). Transportation of farm inputs was small, accounting for 5% of the total impacts.

Substances contributing to this impact category varied. However, they could be divided into two main groups; (i) heavy metals and (ii) pesticides. The contribution of heavy metals (approximately 70% of the total impacts) was larger than that of pesticides (approximately 30% of the total impacts). The main heavy metals included Zn, Cu and Cr, and the pesticides included herbicides and insecticides.

The release of heavy metals to water at the stage of manufacturing of chemical fertilisers was responsible largely for this impact category, followed by heavy metals released during the use of chemical fertilisers, especially P fertilisers. The main sources of pesticides were those used in the production of maize grain (ingredient for concentrate feeds) and maize silage.

It has been reported that substances contributing to this indicator regarding agricultural products vary (e.g. Berthoud et al. (2011) for wheat and Yang (2013) for silage maize), depending on levels and sources of those substances. Yang (2013) reported that pesticides were the major contributor in the life cycle of ethanol produced from ensilaged maize. This is due to the fact that large amounts of pesticides were used during the growing of maize. In contrast, pesticide use was minor in this case study.

3.3.3 Sensitivity analysis

Sensitivity analysis was performed to investigate potential changes in results due to methodological choices, e.g. model parameters (source of input data), inventory models, key assumptions, and impact assessment methods. In the present study, sensitivity analysis was focused on choice of (i) heavy metal contents in chemical fertilisers (i.e. different sources of data for heavy metals content of chemical fertilisers), (ii) assumptions regarding the fate of pesticides, and (iii) time frames for the impact on CC (i.e. 20, 100 and 500 years). The sensitivity analyses were run using average data derived from the 53 dairy farms.

3.3.3.1 Heavy metal content in chemical fertilisers

This sensitivity analysis focused on the three impacts categories whose results were dominated by the effects of heavy metals: Cancer, Non-cancer, and Ecotox. Two alternative data sources were used to calculate the quantities of heavy metals in chemical fertilisers used in the dairy systems. The scenario where the concentrations of heavy metals were derived from the ecoinvent database (ecoinvent scenario) had higher impacts on Cancer (Figure 3.3a), Non-cancer (Figure 3.3b) and Ecotox (Figure 3.3c) relative to the NZ-specific measured concentrations (current scenario) by 86%, 119% and 17% respectively. The corresponding values for these impacts using data derived from the Agric-Footprint database (Agric-Footprint scenario) were higher than the current scenario by 127%, 111% and 16% respectively. These differences were due to the substantially lower heavy metal contents in the current scenario compared with the other two data sources, especially for Cr which has a high characterization factor for the impact on cancer effects, and Zn which has a high characterization factor for the impact on Non-cancer effects. Therefore, as toxicity-related impact indicators in the cradle-to-farm gate life cycle of pasture-based milk are sensitive to concentrations of heavy metals, especially for chemical fertilisers which are the major source of heavy metals, it is recommended that reliable measured data sources should be used.

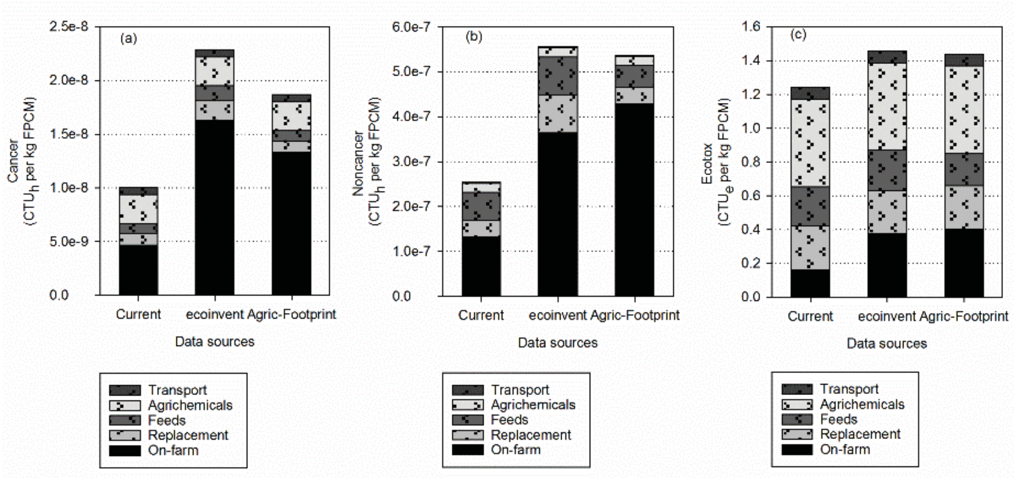

Figure 3.3 Sensitivity analysis of effects of choice of data sources (heavy metal contents in chemical fertilisers) between the NZ-specific (measured data derived from a major fertiliser company in New Zealand), ecoinvent (data derived from the Ecoinvent database) and Agric-Footprint (data derived from the Agric-Footprint database) scenarios on (a) Cancer, (b) Non-cancer and (c) Ecotox impact category results for the cradle-to-farm gate life cycle of milk in the Waikato region, New Zealand for the year 2010/11.

3.3.3.2 Fate of pesticides

This sensitivity analysis on the fate of pesticides was focused on the Ecotox impact category as pesticides were the dominant contributor to this indicator. The pesticide fate method used in the current study (current scenario) was compared with a scenario where it was assumed that all pesticide active ingredients were only deposited onto agricultural soils (soil-only scenario). The Ecotox result was 8% lower for the soil-only scenario compared with the current scenario (Figure 3.4). This reduction was mainly from the decreased contributions of pesticides used at the on-farm stage (8%), production of brought-in feeds (26%) and the production of concentrate feeds for use in the stage of rearing of replacement animals (12%). This was due to the fact that characterization factors of most pesticides released to agricultural soils (i.e. the fate of pesticides applied in the soil-only scenario) in this impact category were lower than when they were released to the other two environmental compartments (i.e. air and water) (Rosenbaum et al. 2008). The findings confirmed that more accurate and precise modelling associated with the fate of pesticides is important (Rosenbaum et al. 2015).

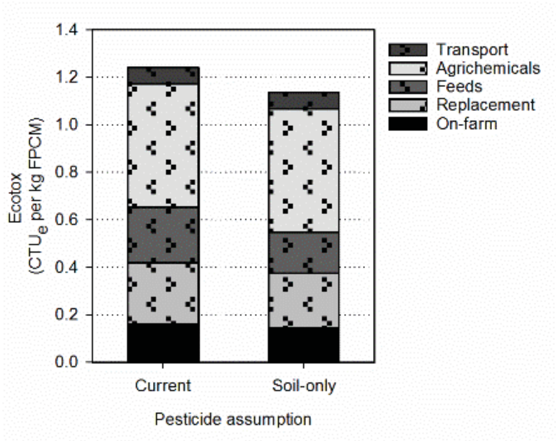

Figure 3.4 Sensitivity analysis of effects of choice of assumption associated with the fate of pesticides on the impact on Ecotox between the current (soil, air and water) and soil-only scenarios in the cradle-to-farm gate life cycle of milk in the Waikato region, New Zealand for the year 2010/11.

3.3.3.3 Time frames in the impact on Climate Change

The sensitivity analysis showed that the CC results were 1.85, 0.83 and 0.35 kg CO_2 eq. per kg FPCM for the time horizons of 20, 100 (i.e. as used in the current study) and 500 years, respectively (Figure 3.5a). As expected, the change in contribution of CH_4 was largely responsible for this change (Figure 3.5b) since it was the biggest component. The results highlighted that it is important to choose the most appropriate time horizon for the decision situation supported by the LCA study (Reisinger and Ledgard 2013).

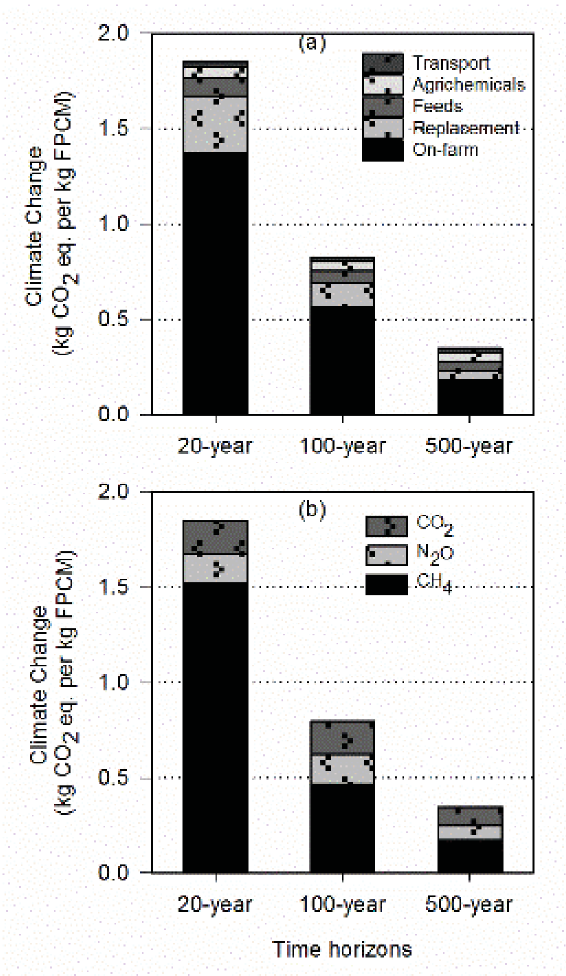

Figure 3.5 Sensitivity analysis of effects of choice of different time-horizons (i.e. 20, 100 and 500 years) on Climate Change impact category for the cradle-to-farm gate life cycle of milk in the Waikato region, New Zealand for the year 2010/11, for (a) the contribution of life cycle stage and (b) the contribution of greenhouse gases.

3.4 Conclusions

In the present study, an attributional LCA was used to assess the environmental profiles of the cradle-to-farm gate life cycle of 53 individual dairy farms. A contribution analysis was performed to identify environmental hotspots. It was found that the on-farm stage contributed 52-73% of the total indicator results for CC, Non-cancer, PM, POFP, AP, TEP and MEP. Focusing on emissions at the on-farm stage, the contribution of enteric CH_4 emissions dominated the CC result, while other on-farm activities differently contributed to different impact categories; for example, the emission of heavy metal (Cr) resulting from the use P fertilisers was mainly responsible for the Ecotox result. In addition, on-farm NH_3 and NO_3 emissions contributed to the PM and

AP, and MEP indicator results, respectively, with all originating mainly from animal excreta and the use of N fertilisers. On-farm NO_x emissions released from the use of fossil diesel dominated the POFP results.

As noted in Section 3.1, the majority of LCAs on dairy farming systems have studied the CC impact category. This leads to a focus on on-farm activities for improvement options because on-farm activities generally make the greatest contribution to the CC result. However, the present study indicates that, when the LCA is expanded to consider a wide range of impact categories, more attention should be given to improvement options related to off-farm activities (i.e. production and transport of agrichemicals and animal feeds, and rearing of replacement animals). This is because these off-farm activities contribute a higher proportion of the result for all other impact indicators than is observed for the CC indicator, with the exception of AP, TEP and MEP. These off-farm contributions ranged from 36% for the POFP to 94% for the IR indicator result. In particular, more than 50% of the total indicator result was contributed by off-farm activities for the ODP, Cancer, IR, FEP and Ecotox impact indicators. For these indicators, the manufacturing of agrichemicals made the largest contribution to the off-farm indicator results (i.e. as opposed to off-farm transport, production of animal feeds or rearing of replacement animals).

As previously mentioned, New Zealand dairy farming systems rely largely on grazed pastures but, over recent years, there has been more use of off-farm inputs, e.g. brought-in feeds (Ledgard and Boyes 2013) and N fertilisers (Parfitt et al. 2012). The present study shows that assessment of just CC may be misleading when considering the environmental hotspots associated with these more intensive dairy farming systems because off-farm activities contributed >50% of the results for 5 out of 12 the impact categories assessed in this study (and almost 50% in a further two impact categories: Non-cancer and PM) whereas they contributed only 32% of the CC result.

However, it should be noted that, as for any LCA study, any final assessment concerning the relative overall environmental significance of on-farm versus off-farm activities is dependent upon normalization and weighting of the characterized Impact Assessment results (which may be undertaken either explicitly or implicitly in a subsequent decision-making process). It is also worth noting that these environmental impacts and hotspots in the life cycle of milk were quantified using an attributional

LCA based on the status quo. The environmental impacts associated with future changes in dairy farming systems (e.g. in response to a demand for more milk) may lead to different conclusions about the relative contribution of on-farm and off-farm activities to the comprehensive environmental impacts associated with the life cycle of milk.

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Supplementary material B

Table B1 Primary inputs used to compute environmental impacts of input products in the off-farm stage (per hectare).

Inputs	Units	Maize silage	Maize grain	Barley grain	Pasture silage	Pasture hay
1. Seeds	kg	32	90	135	-	-
2. Chemical fertilisers						
2.1 Nitrogen (N) [a]	kg N	183	145	98	25	25
2.2 Phosphorus (P) [b]	kg P	38	23	15	6	7
2.3 Lime	kg	500	500	500	100	100
3. Pesticides						
3.1 Glyphosate	kg ai [c]	0.81	0.91	2.7	-	-
3.2 Pesticides	kg ai	3.12	3.89	2.2	-	-
4. Plastic sheet	kg	8	-	-	12	-
5. Diesel	kg	168	111	72	26	21
6. Electricity	kWh	3.91	3.91	3.91	-	-
7. Yields	kg DM	23,000	10,235	5,874	2,500	2,500

[a] urea was used as the N fertiliser for all product systems; [b] diammonium phosphate and single superphosphate were used for crop and pasture production, respectively; [c] ai = active ingredient.

Table B2 Emission models/factors used to quantify specific emissions in dairy farming systems.

	Emission factor/models	Units	Sources
Methane (CH_4)			
• Enteric fermentation [a]	0.065 × total *GEI*/55.65	kg CH_4	MPI (2013)
• Faecal emissions [b]	0.00098198 × on-pasture *FDM*	kg CH_4	MPI (2013)
• Farm dairy effluent (*FDE*)	0.06397826 × effluent *FDM*	kg CH_4	MPI (2013)
Nitrous oxide (N_2O)			
• Urine N deposited onto pastures [c]	0.0116 × urinary N	kg N_2O-N/kg urinary N	Kelliher et al. (2014)
• Faecal N deposited onto pastures [c]	0.0023 × faecal N	kg N_2O-N/kg faecal N	Kelliher et al. (2014)
• Application of *FDE*	0.01 × *FDE* N applied	kg N_2O-N/kg effluent N	MPI (2013)
• Nitrogen (N) fertilisers	0.01 × fertiliser N	kg N_2O-N/kg fertiliser N	MPI (2013)
• Leached nitrogen	0.0075 × leached N	kg N_2O-N/kg NO_3-N	MPI (2013)
• Deposition of volatilized NH_3	0.01 × total NH_3-N loss	kg N_2O-N/kg NH_3-N	MPI (2013)
Nitrogen Oxides (NO_x)			
• On-farm NO_x emissions	0.21 × N_2O (kg)	kg NO_x	Nemecek and Schnetzer (2011)
• Silage NO_x loss	0.5 × total N-silage loss	kg NO_x	Del Prado et al. (2011)
Ammonia (NH_3)			
• Animal excreta [d]	0.1 × total on-pasture N excreted	kg NH_3-N/kg nitrogen	MPI (2013)
• Farm dairy effluent	0.1 × total effluent N	kg NH_3-N/kg nitrogen	MPI (2013)
• Nitrogen fertilisers	0.1 × fertiliser N	kg NH_3-N/kg nitrogen	MPI (2013)
• Silage	0.5 × total N-silage loss	kg NH_3-N	Del Prado et al. (2011)
Nitrate (NO_3)			
• Leached nitrogen	*Overseer*	kg NO_3-N/ha/y	Wheeler et al. (2003)
• Carbon dioxide (CO_2)			
• Urea application	0.773	kg CO_2/kg urea	IPCC (2006)
• Lime application	0.412	kg CO_2/kg lime	IPCC (2006)
• Combustion of fossil diesel (LCA)	3.12	kg CO_2 eq./kg diesel	Nemecek and Kägi (2007)
• Combustion of petrol	3.00	kg CO_2 eq./kg petrol	Nemecek and Kägi (2007)
Phosphorus (P)			
• P losses	*Overseer*	kg P/ha/y	Wheeler et al. (2003)
Volatile Organic Compounds (VOCs)			
• On-pasture manure	28.71	g VOCs/kg DM manure	Rotz et al. (2013)
• Stored manure	60.06	g VOCs/kg DM manure	Rotz et al. (2013)
• Maize silage	34.18	mg VOCs/kg DM silage	Rotz et al. (2013)
• Grass silage	44.07	mg VOCs/kg DM silage	Rotz et al. (2013)

[a] *GEI*: gross energy intake, gross energy contents of feed were estimated using energy factors 23.4 MJ/kg protein, 39.7 MJ/kg fat and 17.4 MJ/kg carbohydrates (Waghorn 2007). Chemical composition of feed were derived from Kolver (2000). [b] *FDM*: faecal dry matter output, estimated proportion of faecal dry matter deposited onto pastures was based on Ledgard and Brier (2004). [c] Estimated proportion of urinary N: faecal N is based on Luo and Kelliher (2010). [d] Estimated total N in animal excreta was computed using a balance method: N excreta = N intake – N retention (milk, weight gain and foetus) (MPI 2013).

Table B3 Primary data used to estimate environmental impacts of feeds used and replacement animals in dairy farming systems.

Product	Units [a]	CC	USEtox impacts			ODP	PM	IR	POF	AP	EP		
			Cancer	Non-cancer	Ecotox.						Terrestrial	Freshwater	Marine
Feed [b]													
- Maize silage	kg DM	0.14E+00	1.24E-08	3.41E-07	4.69E+00	1.27E-08	2.34E-04	6.98E-03	1.80E-03	7.26E-03	3.19E-02	1.03E-04	1.41E-03
- Pasture silage	kg DM	0.23E+00	4.48E-08	3.48E-06	0.94E+00	1.67E-08	3.38E-04	1.63E-02	1.22E-03	9.22E-03	3.89E-02	7.55E-05	3.04E-03
- Pasture hay	kg DM	0.23E+00	2.35E-08	6.12E-07	1.23E+00	1.29E-08	2.70E-04	1.79E-02	1.16E-03	5.38E-03	2.09E-02	8.10E-05	2.90E-03
- Maize grains	kg DM	0.30E+00	2.16E-08	4.76E-07	1.08E+01	2.72E-08	3.40E-04	1.60E-02	1.30E-03	8.59E-03	3.55E-02	2.03E-04	1.31E-03
- Barley grains	kg DM	0.33E+00	5.73E-08	4.67E-06	1.88E+00	2.39E-08	4.09E-04	1.50E-02	1.38E-03	1.04E-02	4.30E-02	2.46E-04	1.97E-03
- Palm kernel expeller [c]	kg DM	0.57E+00	3.85E-09	2.69E-07	0.28E+00	2.59E-09	1.29E-04	1.46E-03	5.06E-04	4.88E-03	2.00E-02	9.64E-05	2.47E-03
Replacements [d]													
- Holstein-Friesian (HF)	kg BW	7.16E+00	1.09E-07	4.74E-06	1.46E+01	9.27E-08	4.18E-03	1.31E-01	2.04E-02	1.56E-01	6.81E-01	1.05E-03	2.03E-02
- Jersey × HF	kg BW	8.05E+00	1.19E-07	5.20E-06	1.57E+01	1.01E-07	4.65E-03	1.44E-01	2.26E-02	1.75E-01	7.61E-01	1.15E-03	2.32E-02

[a] DM: dry matter, BW: body weight. [b] All feed types were produced in New Zealand, except palm kernel expeller which was assumed to be imported from Malaysia. [c] Data derived from the Agric-Footprint database with modification of releases of pesticides and heavy metals. [d] Farm inputs were derived from Basset-Mens et al. (2009) with modification associated with levels of feed intake, as recommended by DairyNZ (2014).

Table B4 Estimated chemical composition and nutritive values of feed used in New Zealand dairy farming systems (Kolver 2000).

Feed	DM (%)	CP (%DM)	EE (%DM)	Ash (%DM)	NDF (%DM)	GE [a] (MJ/kg DM)	DMD [b] (%)
Grazed pasture	15	23	4.5	11	40	16.25	78
Pasture silage	23	17	3.0	9.9	47	16.17	72
Pasture hay	85	17	2.6	9.0	55	16.24	72
Maize silage	33	8	3.1	4.0	43	17.31	73
Maize grain	89	8	4.3	1.6	9	18.00	80
Barley grain	89	12	2.0	2.8	18	17.25	80
Palm kernel expeller	90	14	8.0	4.1	70	18.33	65

[a] Estimation of gross energy (GE) content of feed was computed using factors: 23.4 MJ/kg proteins with subtraction of non-protein nitrogen by 30%, 39.7 MJ/kg fat, 17.4 MJ/kg carbohydrates; [b] DMD: dry matter digestibility, which was used primarily to estimate faecal excretion.

Table B5 Proxies for common materials/processes used in the production of off-farm inputs and relevant on-farm activities.

Materials/processes	Process proxy/reference
1. Seeds/planting materials	Seeds for sowing (Global), market (ecoinvent)
2. Chemical fertilisers (N-P-K)	
2 .1 Urea (46-0-0)	Urea, as N (Global), market (ecoinvent)
2.2 Ammonium sulphate (21-0-0)	Ammonium sulphate, as N (Europe), production (ecoinvent)
2.3 Diammonium phosphate (18-20-0)	Diammonium phosphate, as N (Europe), production (ecoinvent) Diammonium phosphate, as P_2O_5 (Europe), production (ecoinvent)
2.4 Single superphosphate (0-9-0)	Single superphosphate, as P_2O_5 (Europe), production (ecoinvent)
2.5 Potassium chloride (0-0-50)	Potassium chloride, as K_2O (Global), production (ecoinvent)
2.6 Magnesium oxide (MgO)	Magnesium oxide (Europe), at plant (ecoinvent)
2.7 Lime	Lime (Europe), at plant (Agri-Footprint)
3. Pesticides	
3.1 Glyphosate	Glyphosate (Europe), market (ecoinvent)
3.2 Pesticides (unspecified)	Pesticides unspecified (Europe), market (ecoinvent)
4. Plastic, linear low density polyethylene	Polyethylene, linear low density (Europe), production (ecoinvent)
5. Fossil diesel	Diesel (Europe), regional storage (ecoinvent)
6. Combustion of fossil diesel	Diesel, (Global), burned in building machine
7. Lubricating oil [a]	Lubricating oil (Europe), at plant (ecoinvent)
8. Electricity [b]	Proportional mix of New Zealand electricity generation using process from the ecoinvent database
9. Domestic transport (from retailers to farms)	
9.1 Bulky materials (e.g. fertilisers and lime)	Lorry 16-32 t, EURO3 (Europe) (ecoinvent)
9.2 Other materials (e.g. fossil diesel) lubricating oil, pesticides, plastic)	Lorry 3.5-7.5 t, EURO3 (Europe) (ecoinvent)

[a] Estimation of lubricants used in agricultural machine was based on the fossil diesel: lubricant ratio required by internal combustion of agricultural machinery of 1: 0.022 (ecoinvent). [b] Substances released from generation of New Zealand electricity grid mix were estimated using electricity generation processes in the ecoinvent database as proxies (electricity production). The proportion is referred to the Ministry of Business Innovation and Employment (2013): 58% hydropower (run-of-river), 18% natural gas, 13% geothermal, 5% coal, 4% wind power and 1% bioenergy.

Table B6 Factors used to estimate pesticide active ingredients (a.i.) released to the air, water and agricultural soils.

Compartment	Vapour pressure (mPa) [a]	Fractions of pesticide a.i. (%)	References
Air	mPa > 10	95	Webb et al. (2013)
	1 < mPa < 10	50	
	0.1 < mPa < 1	15	
	0.01 < mPa < 0.1	5	
	mPa < 0.01	1	
Water		5	Kellogg et al. (2002)
Soil [b]		100 – (air + water)	Audsley et al. (2003)

[a] Vapour pressure of individual pesticides was derived from the PPDB (2014). [b] The rest of pesticides were assumed to be deposited onto soils and capped at 85% (Audsley et al. 2003).

Table B7 Fossil energy and electricity used in NZ dairy farming systems. Data were averaged across dairy farming in the North Island, New Zealand.

Energy sources	Units	Values
(1) Electricity	MJ/kg milksolids	2.65
(2) Diesel	MJ/kg milksolids	1.71
(3) Petrol	MJ/kg milksolids	0.80

Source: StatisticsNZ (2014).

DRC 16

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STATEMENT OF CONTRIBUTION TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Jeerasak Chobtang

Name/Title of Principal Supervisor: Sarah McLaren

Name of Published Research Output and full reference:

Chobtang J, Ledgard SF, McLaren SJ, Zonderland-Thomassen M, Donaghy DJ (2016) Appraisal of environmental profiles of pasture-based milk production: a case study of dairy farms in the Waikato region, New Zealand. Int J Life Cycle Assess 21 (3):311-325. doi:10.1007/s11367-016-1033-9

In which Chapter is the Published Work: Chapter 3

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate:

  and / or

- Describe the contribution that the candidate has made to the Published Work:

  The candidate was responsible for developing a research framework, performing environmental assessment, interpreting the results and writing the majority of the manuscript.

Jeerasak Chobtang	12 May 2016
Candidate's Signature	**Date**
Sarah McLaren	16 May 2016
Principal Supervisor's signature	**Date**

GRS Version 3– 16 September 2011

Chapter 4 Attributional Life Cycle Assessment of High and Low Intensification Pasture-Based Milk Production Systems in the Waikato Region, New Zealand

This chapter has been modified and accepted for publication in Journal of Cleaner Production.

Abstract

Intensification of pasture-based dairy farming systems is usually associated with increased use of off-farm inputs to increase milk production per hectare, and additional environmental impacts may be associated with upstream production and delivery of these off-farm inputs. The objective of this study was to compare two levels of dairy farming intensification (i.e. high versus low) in pasture-based milk production systems with respect to (i) farm production and (ii) life cycle environmental indicators. The study considered 53 dairy farms in the Waikato region, New Zealand. Three farm attributes (stocking rate, amount of brought-in feeds and amount of nitrogen fertiliser used) were used to define a level of farming intensification (Intensification Index) for each dairy farm. The upper and lower quartiles of the dairy farms ranked on their Intensification Indices were chosen to represent high and low intensification groups, respectively. Twelve midpoint environmental indicators of the dairy systems were assessed using attributional Life Cycle Assessment with one kg of fat- and protein-corrected milk as a functional unit and a cradle-to-farm gate perspective as a system boundary. Compared with the low intensification group, the high intensification group had (on a per hectare basis) higher (P<0.001) stocking rate, total brought-in feeds and nitrogen fertiliser use, which led to greater milk yield per cow (P<0.01) and per hectare (P<0.001). The different levels of farming intensification did not affect total feed conversion efficiency (P>0.05). However, for the high intensification group, the results for 10 out of 12 environmental indicators per kg of fat- and protein-corrected milk were higher (P<0.05) than those in the low intensification group. The main drivers for the increases in most environmental indicator results were the production of brought-in feeds, manufacturing of agrichemicals and transportation of off-farm inputs for use on dairy farms. In contrast, increased pasture intake was negatively correlated (P<0.05) with all environmental impacts, indicating that efficient pasture management is also critical to mitigate environmental impacts. In conclusion, while an increase in intensification of pasture-based dairy farming systems led to increased milk production per cow and per hectare, it also resulted in increased environmental impacts for most indicators. Apart from increased resource use efficiency, increased pasture utilization efficiency is a promising measure to improve environmental sustainability of pasture-based dairy farming systems.

Keywords Dairy ● Environmental impact ● Intensification ● Life Cycle Assessment ● New Zealand ● Pasture-based system

4.1 Introduction

The increasing global population and rising levels of affluence are driving an increased demand for food. At the same time, increasing environmental problems at both global (e.g. climate change; FAO (2006)) and local (e.g. degradation of waterways; Foote et al. (2015)) scales have focused attention on the realization of more environmentally-friendly food systems (Echeverría et al. 2014). As a consequence, consideration is being given to sustainable intensification of food production systems in order to increase productivity within environmental limits (Byerlee et al. 2014; Tilman and Clark 2014) and to changing food consumption patterns (Hallström et al. 2015).

From an increased productivity perspective, one of the most promising approaches that can contribute to meeting the increasing demand for food products is agricultural (farming) intensification, which is associated mainly with increased use of off-farm inputs (Tilman et al. 2011). However, the increased use of off-farm inputs (e.g. fossil fuels, chemical fertilisers, pesticides and brought-in feeds) to support farm intensification can result in increased total environmental impacts associated with food products (Stoate et al. 2009; Tscharntke et al. 2012).

In the dairy sector in particular, intensification of farming systems is generally interpreted as an increase in milk yield either per unit of farm input (Udo et al. 2011) or per unit of dairy farmland (Crosson et al. 2011). This is achieved partly through using a high animal stocking rate coupled with increased use of off-farm inputs (MacLeod and Moller 2006). The increased use of off-farm inputs contributes additional environmental impacts to the life cycle of milk (Vitousek et al. 2009) and may result in increased net environmental impacts per unit of milk produced (MacLeod and Moller 2006).

In principle, accounting for a range of intensification-defining farming practices and combining them to produce an 'Intensification Index' provides a comprehensive categorization of farming intensification (Bava et al. 2014). To achieve this, a mathematical technique has been recommended using to determine degrees or levels of farm intensification (Legendre and Legendre 1998). By using this technique, a number of farm traits defining farming intensification are averaged to form a single

Intensification Index. This technique has been widely used in ecological studies of agricultural systems, e.g. Allan et al. (2014), Blüthgen et al. (2012) and Herzog et al. (2006) but has not been used to quantify levels of dairy farm intensification.

Life Cycle Assessment (LCA) is a systematic approach used for assessing the potential environmental impacts of a product system over its life cycle (International Organization for Standardization 2006a, 2006b). The approach can be used to assess multiple environmental indicators which can help in avoiding problem-shifting both between different environmental impacts and between different life cycle stages of a product (Finnveden et al. 2009). In addition, contribution analysis can help to identify environmental hotspots along the life cycle of product systems which can facilitate development of mitigation options to reduce environmental impacts (Verbeeck and Hens 2010).

There are a number of studies assessing a single environmental impact category (e.g. carbon footprint) (Flysjö et al. 2011; Dalgaard et al. 2014) or only a few impact categories (Battini et al. 2016; Guerci et al. 2013; Bava et al. 2014) of dairy farming systems. However, to our knowledge, there are no published studies directly comparing multiple environmental indicators derived from different quantitative intensification levels of pasture-based dairy farming systems. Therefore, the objective of the present study was to use LCA to quantify the environmental impacts and identify environmental hotspots in pasture-based milk production systems at two contrasting intensification levels (i.e. high versus low), using data from 53 pasture-based dairy farms in the Waikato region, New Zealand.

4.2 Methods

There were four main steps in this study. Firstly, computation of an 'Intensification Index' indicator for each of the 53 dairy farms was undertaken using three main attributes of dairy farms to represent on-farm intensification in pasture-based dairying systems; stocking rate, total amount of brought-in feeds and total amount of nitrogen (N) fertiliser use (see Section 4.2.1). Secondly, dairy farms were ranked based on their intensification indices and the farms in the lower and upper quartiles were selected to represent low and high levels of farm intensification, respectively (see Section 4.2.2). Thirdly, a set of environmental indicators per kg standardized milk produced for each farm were calculated using an attributional LCA (see Section 4.2.3). Finally, a statistical analysis was carried out in order to compare the impacts between the low and high levels of dairy farming intensification and to investigate relationships between farm attributes and impact categories (see Section 4.2.4).

4.2.1 Development of Intensification Index for defining intensification

In order to investigate the influence of farm intensification level on the environmental impacts associated with pasture-based dairy farming systems, it was necessary to first develop a quantitative indicator to determine the level of intensification on a farm. However, as there is no existing well-defined indicator for measuring levels of farming intensification (Bava et al. 2014), the degree of land-use intensity was used as a proxy, as recommended by Kleijn et al. (2009). In addition, Kleijn et al. (2009) recommended that using farm inputs (e.g. N fertilisers and stocking rates) as indicators is more suitable than using farm outputs (e.g. milk yields) to determine land-use intensity. Therefore, an Intensification Index indicator for each dairy farm was calculated based on three farm inputs identified by other researchers as determining pasture-based on-farm intensification (MacLeod and Moller 2006; Oenema et al. 2014; Pinares-Patiño et al. 2009), i.e. stocking rate (livestock unit per ha), N fertilisers (kg N per ha), and brought-in feeds (kg dry matter (DM) per ha).

It should be noted that one or more of these inputs have been used in a qualitative way in other studies for defining the level of dairy farm intensification (e.g. Basset-Mens et al. (2009); Ledgard et al. (2004)). However, these attributes have not been treated as quantitative indicators.

In order to formulate an Intensification Index, these three farm attributes (each quantified per-ha of effective dairy farmland basis, i.e. land used for feeding/grazing dairy cows, excluding farm lanes and buildings) were individually scaled to a non-unit indicator according to Legendre and Legendre (1998), i.e. a minimum value was subtracted from a measured value, then divided by a range (i.e. maximum value minus minimum value) of the corresponding farm attribute. As a result, the indicator ranged between 0 and 1; where 0 refers to the lowest intensity and 1 to the highest intensity (see Table C1 of the Supplementary material C). The three quantified intensification indicators for individual farms were then averaged (arithmetic mean) to give the result for that farm on the Intensification Index; equal contribution of these farm attributes to dairy farm intensification was assumed.

4.2.2 Ranking dairy farms using an Intensification Index

All of the 53 dairy farms were ranked using this Intensification Index. Dairy farms in the lower and upper quartiles were chosen to represent the low and high levels of farm intensification respectively, resulting in 14 farms for each intensification group (see Table C1).

4.2.3 Life Cycle Assessment of dairy farming systems

An attributional LCA was used to assess environmental impacts of the life cycle of milk for all 53 dairy farms in the Waikato region of New Zealand for the production year 2010/11. Details of general characteristics of these dairy systems and environmental inventory procedures can be found in Chapter 3.

In brief, the LCA methodology described in the International Organization of Standardization standards (14040 and 14044) (International Organization for Standardization 2006a, 2006b) was used. In addition, some dairy-specific methods were adopted as recommended by the International Dairy Federation (2010). In particular, biophysical allocation was used for dairy co-products and economic allocation for feeds, and the functional unit was defined as fat and protein corrected milk (FPCM) where FPCM (kg) = milk yield (kg) × [(0.1226 × fat%) + (0.0776 × protein%) + 0.2534]).

The system boundary was cradle-to-farm gate, meaning that all inputs and outputs starting from the extraction of raw materials through to production of one kg FPCM at the farm gate were accounted for. The exceptions were the emissions associated with farm infrastructure (e.g. buildings, machinery, fences and roads), detergents and animal medicines, due to lack of inventory data.

Twelve out of 15 recommended characterization models (EC-JRC-IES 2011; Hauschild et al. 2013) were selected to use for assessing the environmental impacts (Table 4.1). These impact categories were recommended for food and drink products by the European Food Sustainable Consumption and Production Roundtable (European Food SCP Roundtable 2013). For Climate Change, the more recent characterization factors of the International Panel for Climate Change were used to replace the previous factors, as recommended by Myhre et al. (2013).

Table 4.1 Impact categories selected in the present study, based on the European Food SCP Roundtable (2013). The impacts superscripted by * were not included in the present study due to a lack of relevant inventory data.

Impact category	Abbreviation
Climate Change	CC
Ozone Depletion Potential	ODP
Human Health Toxicity, cancer effects	Cancer
Human Health Toxicity, non-cancer effects	Non-cancer
Particulate Matter	PM
Ionizing Radiation – human health effects	IR
Photochemical Ozone Formation Potential	POFP
Acidification Potential	AP
Eutrophication – terrestrial ecosystems	TEP
Eutrophication – freshwater ecosystems	FEP
Eutrophication – marine ecosystems	MEP
Ecotoxicity for Aquatic Freshwater	Ecotox
Resource Depletion-mineral, fossil *	RD (mineral, fossil)
Resource Depletion-water use *	RD (water use)
Land Use Impact *	LU

A contribution analysis was performed in order to determine environmental hotspots in the cradle-to-farm gate life cycle of milk. The life cycle stages were defined as follows:

- The on-farm stage (On-farm); this stage included emissions associated with the dairy farm system and on-farm activities, e.g. milking cows, use of chemical fertilisers and pesticides, use of fossil fuels, oils and electricity, and management of farm effluent and animal excreta.

- The rearing of replacement animals (Replacement); this stage covered emissions associated with the production of replacement animals (heifers), i.e. from about 4 days of age up to nearly calving. For all farm systems, this stage was carried out off-farm.
- The production of brought-in feeds (Supplements); this stage aggregated emissions associated with the production of brought-in feeds to be used on dairy farms.
- The manufacturing of agrichemicals (Agrichemicals); this stage included emissions originating from the manufacturing of chemical fertilisers and pesticides to be used on dairy farms.
- The transportation (Transport); emissions arising from transportation of brought-in feeds, chemical fertilisers and other off-farm inputs to on-farm were summed in this stage.

4.2.4 Statistical analysis

The main physical farm traits, environmental impacts, and contributions of different life cycle stages for the low and high intensification levels were statistically compared using a group comparison *t*-test analysis (Statistical Analysis Software, SAS® 9.4, SAS Institute). To determine degrees of association between farm attributes and environmental impacts, a correlation analysis was carried out using the CORR Procedure. Data were tested for a normal distribution (using a Shapiro-Wilk test) using the UNIVARIATE Procedure prior to being subjected to a Pearson linear correlation analysis.

4.3 Results and discussion

The following sections compare physical farm traits (Section 4.3.1), potential environmental impacts (Section 4.3.2) and environmental hotspots for each impact category (Section 4.3.3) between low and high levels of dairy farm intensification. Correlation analysis results between individual pairs of environmental indicators, and between key farm traits and environmental indicators are present in Section 4.3.4.

4.3.1 Physical farm traits

The results of the *t*-test analysis for a comparison of physical farm traits between the two contrasting levels of pasture-based dairy farming intensification are presented in Table 4.2.

Table 4.2 A comparison of selected physical farm traits between the low and high intensity pasture-based dairy farms in the Waikato, New Zealand, over one production year (2010/11); a per ha basis refers to the on-farm stage only (i.e. area used for grazing and milking dairy cows).

Farm traits	Units [e]	Overall mean	Intensification levels		SE [g]	Sig. [h]
			Low	High		
(1) Farm number	#	53	14	14		
(2) Stocking rate	LU [f]/ha	2.73	2.37	3.14	0.15	***
(3) Replacement rate	%	23	22	24	1	ns
(4) Fertilisers						
• Nitrogen (N)	kg N/ha	142	50	205	23	***
• Phosphorus (P)	kg P/ha	49	47	57	10	ns
• Potassium (K)	kg K/ha	45	31	50	9	ns
(5) Total dry matter (DM) intake	kg DM/ha	11,300	9,772	13,144	679	***
• Estimated pasture intake	kg DM/ha	9,038	8,869	9,636	506	ns
• Total brought-in feeds offered [a]	kg DM/ha	2,744	1,096	4,250	510	***
○ Palm kernel expeller	kg DM/ha	1,643	481	2,720		
○ Maize silage	kg DM/ha	893	313	1,410		
○ Concentrate feed	kg DM/ha	34	79	0		
○ Pasture silage	kg DM/ha	160	211	107		
○ Pasture hay	kg DM/ha	13	12	14		
(6) % brought-in feed (DM basis) [b]	%	21	11	31	4	***
(7) FCE [c]	kg/kg	1.13	1.11	1.16	0.02	ns
(8) FPCM [d] production	kg/cow	4,730	4,338	5,165	222	**
	kg/ha	12,850	10,959	15,208	908	***

[a] Total brought-in feed intake excluded feed waste during feeding-out, % feed lost was according to DairyNZ (2015); [b] % brought-in feed = (total brought-in feed intake/total

feed intake) × 100; [c] feed conversion efficiency (FCE) was computed as total milk production (kg) per ha/total feed intake (kg dry matter) per ha; [d] Fat and protein corrected milk (FPCM); [e] in all cases, the 'per ha' refers to the dairy farm only and excludes support land for rearing animal replacements and production of brought-in feeds; [f] LU: livestock unit (1 LU = 1 dairy cow at 500 kg body weight), [g] SE: standard error; [h] significant levels derived from a *t*-test comparison between the low and high intensification groups, *** = $P<0.001$, ** = $P<0.01$, * = $P<0.05$, and ns = non-significance ($P>0.05$).

The results in Table 4.2 indicate that, as expected, the main physical farm attributes considered (on a per-ha basis), i.e. stocking rate, on-farm brought-in feeds and on-farm N fertilisers, were higher in the high intensification group ($P<0.001$). On a per-ha basis, although estimated pasture intake between the two groups did not differ ($P>0.05$), the total DM intake was higher for the high intensification group than for the low intensification group ($P<0.001$).

For milk production, Table 4.2 shows that dairy cows in the high group produced more milk per cow ($P<0.01$) and per hectare ($P<0.001$), compared with those in the low group. In contrast, there was no difference in feed conversion efficiency (FCE) between the two contrasting intensity groups ($P>0.05$).

4.3.2 Environmental impacts

The results derived from the *t*-test analysis comparing the environmental impacts computed on a per kg FPCM basis between the two intensity groups are presented in Table 4.3. The estimates for 10 out of 12 environmental indicators in the high intensification group were higher ($P<0.05$) than those in the low intensification group, i.e. Climate Change (CC), Ozone Depletion Potential (ODP), Particulate Matter (PM), IR (Ionizing Radiation), Photochemical Ozone Formation Potential (POFP), Acidification Potential (AP), Terrestrial Eutrophication Potential (TEP), Freshwater Eutrophication Potential (FEP), Marine Eutrophication Potential (MEP) and Ecotoxicity for Aquatic Freshwater (Ecotox). However, the remaining two impact categories (Human Health Toxicity - cancer effects and non-cancer effects) were not different ($P>0.05$) between the two intensification groups.

Table 4.3 Effects of different intensification levels (low versus high) of dairy farming systems in the Waikato region, New Zealand for the production year 2010/11 on environmental impacts (per kg fat- and protein-corrected milk).

Impact categories [a]	Units [b]	Overall mean	Intensification levels		SE [c]	*P*-value [d]
			Low	High		
CC	kg CO_2 eq.	0.80E+00	0.73E+00	0.86E+00	0.03E+00	***
ODP	kg CFC-11 eq.	1.02E-08	0.86E-08	1.16E-08	0.05E-08	***
Cancer	CTU_h	1.00E-08	1.01E-08	1.07E-08	0.01E-08	ns
Non-cancer	CTU_h	2.60E-07	2.70E-07	2.65E-07	0.26E-07	ns
PM	kg $PM_{2.5}$ eq.	4.60E-04	3.91E-04	5.07E-04	0.22E-04	***
IR	kBq U^{235} eq.	1.06E-02	0.89E-02	1.24E-02	0.09E-02	**
POFP	kg NMVOC eq.	2.58E-03	2.32E-03	2.84E-03	0.09E-03	***
AP	molc H^+ eq.	1.53E-02	1.35E-02	1.64E-02	0.06E-02	***
TEP	molc N eq.	6.55E-02	5.82E-02	6.96E-02	0.24E-02	***
FEP	kg P eq.	0.96E-04	0.91E-04	1.08E-04	0.06E-04	*
MEP	kg N eq.	2.67E-03	2.36E-03	2.96E-03	0.14E-03	**
Ecotox	CTU_e	1.23E+00	1.12E+00	1.49E+00	0.10E+00	**

[a] Abbreviations for impact categories refer to as in Table 4.1; [b] CO_2 = Carbon dioxide, eq. = equivalent, CFC-11 = Trichlorofluoro-methane, CTU_h = Comparative toxic unit for humans, $PM_{2.5}$ = Particulate matter less than 2.5 μ in diameter, kBq = kilobecquerel, U^{235} = Uranium-235, NMVOC = Non-methane volatile organic compounds, molc = mole of charge, H^+ = Hydrogen ion, N = Nitrogen, P = Phosphorus and CTU_e = Comparative toxic unit for ecosystems; [c] SE = standard error; [d] significant levels between the low and high intensification groups: *** = $P<0.001$, ** = $P<0.01$, * = $P<0.05$, and ns = non-significance ($P>0.05$).

Based on the literature review, only two LCA studies (Basset-Mens et al. 2009; Bava et al. 2014) have focussed on assessing and comparing environmental impacts between dairy farming systems varying in levels of farm intensification. In New Zealand pasture-based dairy systems, Basset-Mens et al. (2009) reported that intensifying dairy farms through either (i) use of only N fertilisers or (ii) combined use of N fertilisers with brought-in maize silage resulted in higher CC, AP and Eutrophication Potential (EP) than those in a low-input dairy farm when computed on a per kg milk basis. In contrast, in an Italian confinement-based dairying system, Bava et al. (2014) reported that environmental impacts per kg FPCM for CC, AP and EP in more intensive dairying systems did not differ distinctively from the less intensive systems. However, the results from these two studies cannot be directly compared with the results from the present study for several reasons, including the use of different inventory models, impact assessment methods and different methods to classify farming intensification.

4.3.3 Comparison of environmental hotspots

A contribution analysis was performed in order to identify environmental hotspots at the two different intensification levels. In addition, the relative magnitudes of the contributions of the different life cycle stages to each impact category result between the low and high intensification groups were statistically compared, and the results are depicted in Figures 4.1 and 4.2.

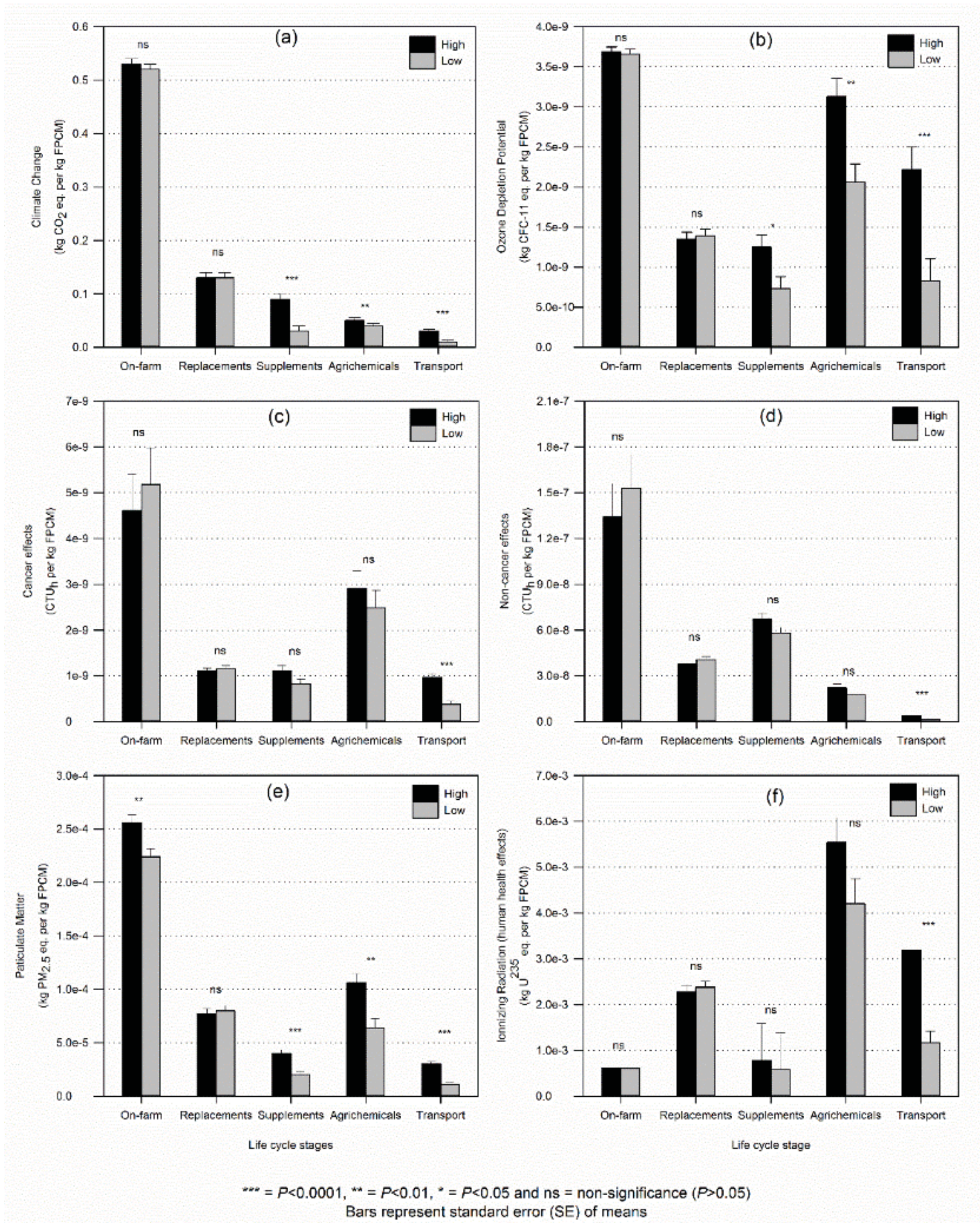

Figure 4.1 A comparison of the contribution of stages in the cradle-to-farm gate life cycle of milk to (a) Climate change, (b) Ozone Depletion Potential, (c) Cancer effects, (d) Non-cancer effects, (e) Particulate Matter and (f) Ionizing Radiation results for low and high intensification levels of dairy farms in the Waikato, New Zealand.

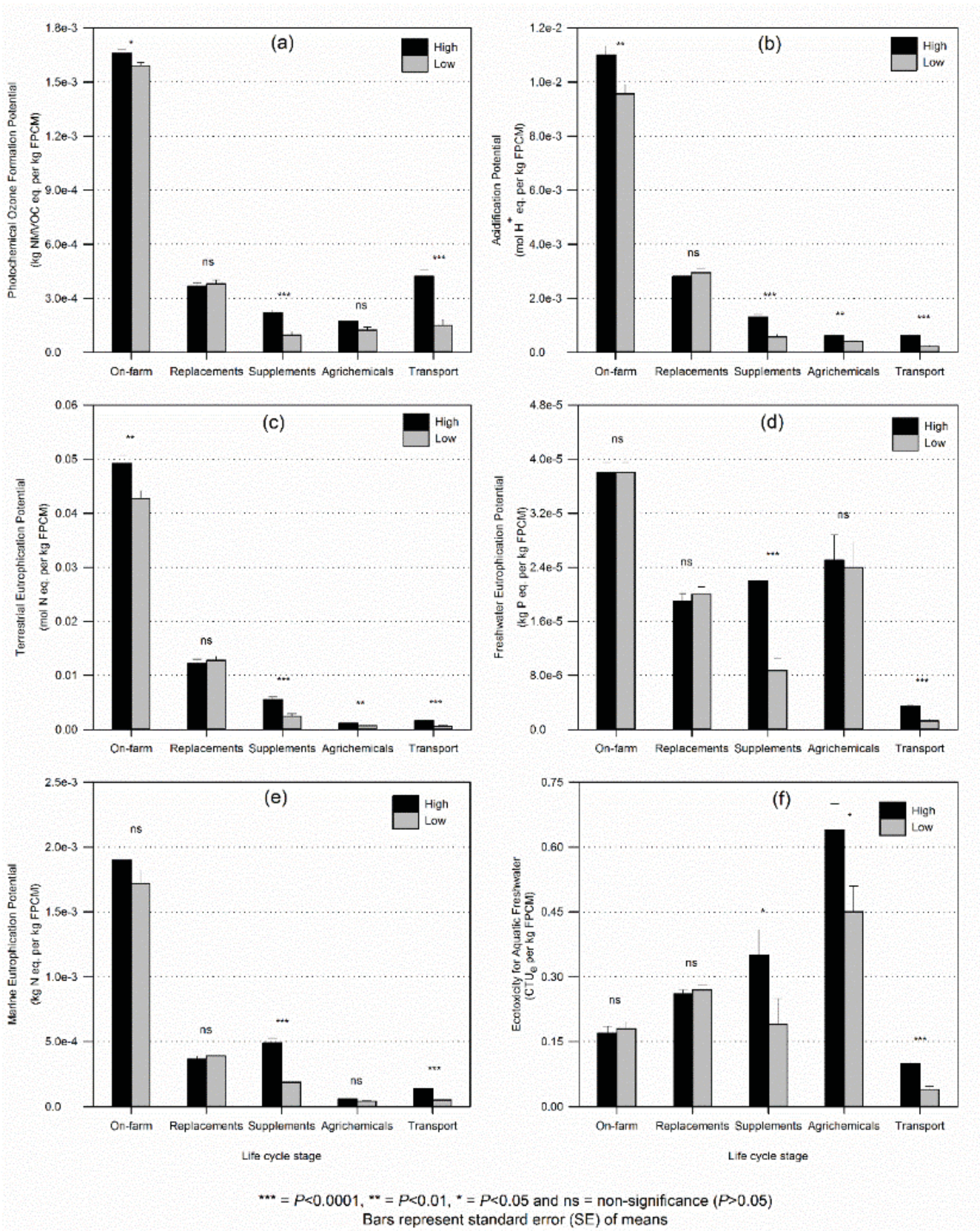

Figure 4.2 A comparison of the contribution of stages in the cradle-to-farm gate life cycle of milk to (a) Photochemical Ozone Formation Potential, (b) Acidification Potential, (c) Terrestrial Eutrophication Potential, (d) Freshwater Eutrophication Potential, (e) Marine Eutrophication Potential and (f) Ecotoxicity for Aquatic Freshwater results for low and high intensification levels of dairy farms in the Waikato, New Zealand.

Overall, the contribution of the on-farm stage was higher ($P<0.05$) for the high intensification group than for the low intensification group for the PM, POFP, AP and TEP impact categories. In contrast, there were no significant differences ($P>0.05$) in any impact categories between the two intensification groups for the contribution of the rearing of replacement animals. The contributions of the production of brought-in feeds, manufacturing of agrichemicals and transportation of off-farm inputs for use on dairy farms were higher ($P<0.05$) for the high intensification group for between 7 and 10 impact categories.

4.3.3.1 Climate Change

Overall, the high intensification group had a higher ($P<0.001$) CC result than the low intensification group (see Table 4.3). Figure 4.1a shows the contribution of the different life cycle stages to the CC impact category. Emissions associated with the on-farm stage and the rearing of animal replacements were not different between the two intensification groups ($P>0.05$). In contrast, the contributions of the production of brought-in feeds, manufacturing of agrichemicals (i.e. chemical fertilisers and pesticides), and transportation of off-farm inputs for use on dairy farms to the CC impact category were all higher ($P<0.01$) for the high intensification group. The main contributing substance associated with these three life cycle stages was carbon dioxide (CO_2) emissions mainly from the use of fossil fuels (data not shown).

4.3.3.2 Ozone Depletion Potential

The ODP result for the high intensification group was higher ($P<0.001$) than for the low intensification group (see Table 4.3). The main drivers of this difference were the (i) production of brought-in feeds, (ii) manufacturing of agrichemicals and (iii) transportation of off-farm inputs for use on a dairy farm (Figure 4.1b). The contributions of these life cycle stages were higher ($P<0.05$) in the high intensification group compared with the low intensification group. The manufacturing of chemical fertilisers and use of fossil fuels for transportation were the major hotspots releasing ozone-depleting substances (e.g. Bromotrifluoro-methane (Halon-1301), bromochlorodifluoro-methane (Halon-1211) and tetrachloro-methane (CFC-10)). However, the contributions of the on-farm stage and the off-farm rearing of replacement animals were not different ($P>0.05$) between the two intensification groups.

4.3.3.3 Human Health Toxicity - cancer effects

There was no effect (P>0.05) of dairy farm system intensification on the Human Health Toxicity (Cancer) results (see Table 4.3). This is due to the fact that there were no differences in the major contributors (life cycle stages) between the high and low intensification groups (P>0.05), except for transportation of off-farm inputs for use on a dairy farm (Figure 4.1c). However, the contribution of the transportation stage was small and therefore had a minimal effect on the total impact.

4.3.3.4 Human Health Toxicity - non-cancer effects

Similar to the Human Health Toxicity (Cancer) results, the Non-cancer results for the low and high intensification groups were not different (P>0.05) (see Table 4.3). Even though the contribution of transportation of off-farm inputs for use on a dairy farm in the high intensification group was greater (P<0.001) than that in the low intensification group, its contribution was minimal and therefore it did not lead to a significant difference in the total Non-cancer results between the two intensification groups (Figure 4.1d).

4.3.3.5 Particulate Matter

The PM results were different (P<0.001) between the two intensification groups (see Table 4.3). The main drivers of this difference (P<0.01) were the (i) on-farm stage, (ii) production of brought-in feeds, (iii) manufacturing of agrichemicals and (iv) transportation of off-farm inputs for use on a dairy farm (Figure 4.1e). Ammonia (NH_3) emissions from the on-farm stage and fine PM from manufacturing of agrichemicals (especially for chemical fertilisers) were the hotspot substances contributing to this impact category result (data not shown). The farms in the high intensification group released more NH_3 and used more N fertilisers than those in the low intensification group. The increased milk yields from the high intensification group could not sufficiently “dilute” pollutants contributing to the impact on PM (especially NH_3).

4.3.3.6 Ionizing Radiation - human health effects

The high intensification group had higher (P<0.01) IR results than the low intensification group (see Table 4.3). The difference was mainly due to the larger contribution from transportation of off-farm inputs for use on a dairy farm (Figure 4.1f). The main contributing substances were C-14 and Ra-222 which were released from operation of transport, mainly associated with production of fossil fuels.

4.3.3.7 Photochemical Ozone Formation Potential

The POFP result for the high intensification group was higher than that of the low intensification group ($P<0.001$) (see Table 4.3). The main life cycle stages contributing to significant differences between the two intensification groups were (i) on-farm stage, (ii) production of brought-in feeds and (iii) transportation of off-farm inputs for use on a dairy farm (Figure 4.2a). The main contributing substances from the on-farm stage were volatile organic compounds (mainly from animal excreta) and methane (CH_4) (mainly from enteric fermentation), while emissions of nitrogen oxides (NO_x) were associated with the production of brought-in feeds and transportation of off-farm inputs for use on a dairy farm (mainly from the use of fossil fuels).

4.3.3.8 Acidification Potential

The high farming intensification group had a higher ($P<0.001$) AP result than the low intensification group (see Table 4.3), and the largest contributor was the on-farm stage (Figure 4.2b). On-farm NH_3 emissions were the largest single substance contributing to this impact category, which was associated mainly with animal excreta (particularly animal urine) and N fertilisers. In the production of brought-in feeds, the use of N fertilisers was the main source of NH_3 emissions. The contributions of manufacturing of agrichemicals and transportation of off-farm inputs for use on a dairy farm were negligible, but higher ($P<0.01$) for the high intensification group than for the low intensification group.

4.3.3.9 Terrestrial Eutrophication Potential

The high intensification group had higher ($P<0.001$) TEP results at all life cycle stages apart from the off-farm rearing of animal replacements. Similar to AP, the on-farm stage had the largest impact on TEP (Figure 4.2c). The single largest emission contributing to this impact category was NH_3 originating mainly from animal excreta (especially for urine patches) and N fertilisers, collectively accounting for 95% of the total impacts.

4.3.3.10 Freshwater Eutrophication Potential

The high intensification group had a higher FEP result than the low intensification group ($P<0.05$). The two main stages that made a difference between the two intensification groups were the production of brought-in feeds and transportation of off-farm inputs for use on a dairy farm (Figure 4.2d). In the production of brought-in feeds, phosphorus loss through runoff was the major contributor. For transportation of off-farm inputs for use on a dairy farm, the contribution was negligible, but higher (P<0.001) in the high intensification group than that in the low intensification group.

4.3.3.11 Marine Eutrophication Potential

The MEP result for the high intensification group was higher than that for the low intensification group (P<0.01). The production of brought-in feeds and transportation of off-farm inputs for use on a dairy farm were the two main life cycle stages (Figure 4.2e), driving higher total impact in the high intensification group than the low group (see Table 4.3). Nitrate (NO_3), NH_3 and NO_x were the major eutrophying substances contributing to the impact on MEP. In the production of brought-in feeds, NO_3 leaching was the largest contributor, followed by NO_x emissions (from use of fossil fuels), with NH_3 playing a minor role. The contribution of transport to this impact category was through NO_x emissions originating mainly from the use of fossil fuels.

4.3.3.12 Ecotoxicity for Aquatic Freshwater

The Ecotox result for the high intensification group was higher (P<0.01) than that in the low intensification group (see Table 4.3). Figure 4.2f shows that this significant difference was influenced largely by off-farm stages. The contributions of the manufacturing of agrichemicals (especially phosphate fertilisers), production of brought-in feeds and transportation of off-farm inputs for use on a dairy farm in the high intensification group were higher (P<0.001) than those in the low intensification group. The major contributing substances were heavy metals (especially zinc, copper and chromium) and pesticides (especially insecticides).

4.3.4 Results of correlation analysis

A correlation analysis was carried out to determine the degrees of association (correlation coefficients, r) between impact categories (Table 4.4) and between selected farm attributes and the impact categories (Table 4.5). The degrees of association of the parameters investigated varied, ranging from weak (r<0.50) to moderate (0.50<r<0.80) and strong (r>0.80).

The CC indicator was positively correlated with all the other indicators investigated (P<0.05), except the Non-cancer indicator (P>0.05) (Table 4.4). Likewise, significant relationships between other individual indicators were obtained (P<0.05), except the Cancer vs TEP and MEP indicators, and Non-cancer vs AP, TEP and MEP indicators which were not significant (*P*>0.05). The Cancer and Non-cancer indicators were highly correlated (*P*<0.05) as they were both driven mainly by release of heavy metals. Similarly, the NH_3-driven indicators (i.e. AP, PM and TEP) were strongly correlated (*P*<0.05).

Table 4.4 Correlation coefficients between impacts categories (per kg fat- and protein-corrected milk (FPCM)) of dairy farming systems in the Waikato region, New Zealand over one production year (2010/11).

	CC	ODP	Cancer	Noncancer	PM	IR	POFP	AP	TEP	FEP	MEP
CC [a]	-										
ODP	0.84[b]										
Cancer	0.37	0.57									
Non-cancer	0.25	0.48	0.93								
PM	0.88	0.87	0.51	0.40							
IR	0.76	0.90	0.85	0.74	0.83						
POFP	0.94	0.90	0.41	0.29	0.78	0.78					
AP	0.88	0.76	0.29	0.20	0.96	0.67	0.72				
TEP	0.84	0.70	0.23	0.15	0.94	0.60	0.67	0.99			
FEP	0.67	0.79	0.85	0.72	0.65	0.92	0.75	0.48	0.41		
MEP	0.71	0.55	0.17	0.10	0.64	0.47	0.64	0.66	0.65	0.51	
Ecotox	0.55	0.82	0.73	0.62	0.68	0.84	0.66	0.51	0.46	0.83	0.31

[a] See Table 4.1 for definition of abbreviation; [b] significant values (*P*<0.05) are underlined, and a value with a double underline refers to a significantly strong relationship (*r*>0.80).

Table 4.5 The correlation coefficients between environmental indicators (per kg FPCM) and farm attributes of dairy farming systems in the Waikato region, New Zealand over one production year (2010/11).

	CC [a]	ODP	Cancer	Non-can	PM	IR	POFP	AP	TEP	FEP	MEP	Ecotox
SR [b]	0.20	0.19	-0.19	-0.23	0.10	0.01	0.29 [c]	0.12	0.12	0.01	0.23	0.11
FPCM/ha	-0.06	0.15	-0.12	-0.16	-0.04	0.00	0.08	-0.12	-0.14	-0.09	-0.06	0.06
Pasture DMI	-0.33	-0.32	-0.29	-0.31	-0.22	-0.34	-0.35	-0.18	-0.16	-0.44	-0.21	-0.30
Supplement DMI	0.51	0.67	0.09	0.05	0.34	0.42	0.73	0.25	0.20	0.43	0.36	0.48
Total DMI	0.05	0.14	-0.19	-0.23	0.02	-0.03	0.16	0.00	0.00	-0.10	0.05	0.04
FCE	-0.29	0.14	0.12	0.13	-0.16	0.08	-0.12	-0.34	-0.39	0.00	-0.28	0.09
N fertiliser	0.71	0.78	0.18	0.09	0.84	0.60	0.67	0.82	0.80	0.36	0.57	0.50
P fertiliser	0.23	0.45	0.91	0.81	0.36	0.73	0.32	0.14	0.07	0.74	0.05	0.66

[a] See Table 4.1 for definition of abbreviation; [b] SR = stocking rate, DMI = dry matter intake, FCE = feed conversion efficiency; [c] significant values (*P*<0.05) are underlined, and a value with a double underline refers to a significantly strong relationship (*r*>0.80).

Significant correlations between some environmental indicators and farm attributes were observed either positively or negatively ($P<0.05$) (Table 4.5). For example, pasture intake was weakly but negatively correlated with all environmental indicators ($P<0.05$), except the PM, AP and TEP indicators ($P>0.05$). In contrast, most environmental indicators (e.g. CC, ODP, PM, IR, POFP, FEP, MEP and Ecotox) were positively correlated with level of intake of feed supplements ($P<0.05$). In confinement-based dairy systems in Italy, Battini et al. (2016) reported that FCE was negatively correlated with the CC indicator result.

While the FCE was negatively correlated with the CC, AP, TEP and MEP indicators ($P<0.05$), the on-farm N and P fertilisers had positive correlations with a number of environmental indicators (see Table 4.5), especially for the relationships between on-farm N fertilisers with PM, AP and TEP as well as those for on-farm P fertilisers with Cancer and Non-cancer where strong correlations were obtained ($r>0.8$). Stocking rate had a significant but weak correlation with only POFP ($P<0.05$).

The results from the correlation analysis highlighted two main interesting aspects. Firstly, there were positive relationships between most pairs of impact categories examined, implying that any intervention in New Zealand pasture-based dairy farming systems that can reduce effects in one impact category can potentially decrease effects in other impact categories (see Table 4.4). It is worth noting that strong correlation coefficients between some impact indicators have also been found in a wide range of other products (Berger and Finkbeiner 2011; Laurent et al. 2012; Röös et al. 2013).

Secondly, there were either positive or negative relationships between impact categories and dairy farm attributes. A positive relationship implies that an increase in a particular impact can result from an increase in the correlated farm attribute(s). In contrast, a negative correlation occurs when an increase in a particular impact was associated with a decrease in the correlated farm attributes (see Table 4.5).

A negative correlation between the quantity of pasture intake and a number of environmental indicators suggested that increased pasture intake is associated with decreased environmental impacts for the cradle-to-farm gate life cycle of pasture-based milk. This is in contrast to an increase in supplementary feed intake which was associated with increased environmental impacts. Furthermore, it should be noted that increased use of brought-in supplementary feed to support dairy farm intensification may be linked with (direct and indirect) land use change, which can potentially further exacerbate some environmental impact categories, e.g. CC (Flysjö et al. 2012; Battini et al. 2016). However, this aspect was not addressed in the present study since it was an attributional LCA study of pre-existing dairy systems, as opposed to potential future dairy systems.

Feed conversion efficiency was negatively correlated with most environmental indicators, although only the impacts on CC, AP, TEP and MEP were strongly significant. This confirms the findings of Guerci et al. (2013), Henriksson et al. (2011) and Hermansen and Kristensen (2011) who reported that FCE is a key factor negatively correlated with environmental impacts (e.g. greenhouse gases and AP) in dairy production systems. In addition, Connor (2015) reported that improved FCE in dairy farming systems led to lowered GHG emissions and environmental impacts related to emissions of manures when computed on a per unit of milk produced. This is due to the fact that less feed is needed per kg of product (i.e. milk) and therefore less excreta are generated. Similarly, Cela et al. (2014) found that increased FCE on dairy farms resulted in a reduction of nutrient surplus (e.g. N and P) per unit of milk produced.

Positive correlations between fertilisers (i.e. N and P fertilisers) and a wide range of environmental indicators were observed (see Table 4.5), implying that increased use of these two fertilisers could increase environmental impacts associated with the life cycle of pasture-based milk production systems. It has been reported that improved fertiliser use efficiency can help reduce the environmental impacts of dairy systems, especially N fertilisers (Gerber et al. 2014).

4.4 Conclusions

This study investigated the differences in productivity and environmental impacts between low and high intensification levels of pasture-based dairy farming systems in the Waikato region, New Zealand. Based on the results, it can be concluded that the high intensification farming systems produced more milk (both per cow and per ha) than the low intensification systems. However, these benefits were at the expense of increases in environmental impacts per kg FPCM for 10 out of 12 environmental indicators investigated. The main drivers for the differences were associated with the production of brought-in feeds, manufacturing of agrichemicals, and transportation of off-farm inputs for use on dairy farms.

The results highlight that pasture-based dairy farm intensification options should focus firstly on increasing pasture intake (maximized pasture utilization efficiency) in order to avoid use of additional brought-in supplementary feeds. Secondly, when off-farm inputs (e.g. brought-in supplementary feed and fertilisers) are used, increased FCE of brought-in feeds and optimizing fertiliser use for production of all feed sources (off-farm and on-farm feed resources) will also be critical in reducing the multiple environmental impacts associated with pasture-based milk production systems.

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Supplementary material C

Table C1 Selected farm attributes defining pasture-based farming intensification (stocking rate (SR, Livestock Unit (LU) per ha), total brought-in feeds (Feeds, kg dry matter per ha), and nitrogen fertiliser (N fertiliser, kg N per ha)), scaled indicators, Intensification Index (Index) and assigned quartiles (i.e. lower and upper quartiles) of 53 dairy farms in the Waikato region, New Zealand over one production year (2010/11). The 'per ha' refers to dairy farm area only, i.e. the area used for grazing of dairy cows.

Farm	SR	Feeds	N fertiliser	Scaled SR	Scaled feeds	Scaled N	Index	Assigned quartile
1	1.90	533	0	0.01	0.09	0.00	0.03	Lower
2	1.90	285	23	0.01	0.05	0.09	0.05	Lower
3	2.19	472	0	0.15	0.08	0.00	0.06	Lower
4	1.88	356	84	0.00	0.06	0.33	0.13	Lower
5	2.60	0	30	0.34	0.00	0.12	0.15	Lower
6	2.40	270	75	0.24	0.04	0.29	0.19	Lower
7	2.18	382	120	0.14	0.06	0.47	0.22	Lower
8	2.28	1,394	75	0.19	0.23	0.29	0.24	Lower
9	2.20	3,966	0	0.15	0.64	0.00	0.26	Lower
10	2.27	2,417	65	0.18	0.39	0.25	0.28	Lower
11	3.45	774	0	0.73	0.13	0.00	0.29	Lower
12	2.67	1,182	80	0.37	0.19	0.31	0.29	Lower
13	2.84	1,429	55	0.45	0.23	0.22	0.30	Lower
14	2.48	1,886	91	0.28	0.30	0.36	0.31	Lower
15	2.36	2,058	118	0.22	0.33	0.46	0.34	
16	2.69	2,785	53	0.38	0.45	0.21	0.35	
17	2.72	857	150	0.40	0.14	0.59	0.37	
18	2.56	1,729	135	0.32	0.28	0.53	0.37	
19	2.82	1,165	127	0.44	0.19	0.50	0.38	
20	2.28	2,293	154	0.19	0.37	0.60	0.39	
21	2.74	954	157	0.40	0.15	0.62	0.39	
22	3.04	1,098	120	0.54	0.18	0.47	0.40	
23	2.24	884	234	0.17	0.14	0.92	0.41	
24	2.83	2,177	119	0.44	0.35	0.47	0.42	
25	2.99	2,059	105	0.52	0.33	0.41	0.42	
26	2.88	1,784	138	0.47	0.29	0.54	0.43	
27	2.23	2,624	182	0.16	0.42	0.71	0.43	
28	2.85	2,898	120	0.45	0.47	0.47	0.46	
29	2.69	2,551	154	0.38	0.41	0.60	0.47	
30	2.64	1,792	197	0.36	0.29	0.77	0.47	
31	2.88	1,958	170	0.47	0.32	0.67	0.48	
32	2.84	3,831	106	0.45	0.62	0.42	0.49	
33	2.91	2,650	148	0.48	0.43	0.58	0.50	
34	2.74	2,754	175	0.40	0.44	0.69	0.51	
35	2.61	3,744	162	0.34	0.60	0.64	0.53	
36	3.11	3,279	130	0.57	0.53	0.51	0.54	
37	2.61	2,465	227	0.34	0.40	0.89	0.54	

38	2.52	3,297	205	0.30	0.53	0.80	0.55	
39	3.38	2,438	153	0.70	0.39	0.60	0.56	
40	2.72	3,300	197	0.39	0.53	0.77	0.57	Upper
41	2.92	4,265	179	0.49	0.69	0.70	0.63	Upper
42	3.17	3,634	185	0.60	0.59	0.73	0.64	Upper
43	3.03	3,192	227	0.54	0.52	0.89	0.65	Upper
44	2.93	4,150	200	0.49	0.67	0.78	0.65	Upper
45	2.88	3,114	255	0.47	0.50	1.00	0.66	Upper
46	2.83	3,680	237	0.44	0.59	0.93	0.66	Upper
47	3.33	4,114	196	0.68	0.66	0.77	0.70	Upper
48	3.81	3,276	183	0.90	0.53	0.72	0.72	Upper
49	2.92	4,381	244	0.49	0.71	0.96	0.72	Upper
50	3.07	4,957	207	0.56	0.80	0.81	0.72	Upper
51	3.35	6,030	142	0.69	0.97	0.55	0.74	Upper
52	2.94	5,217	254	0.50	0.84	1.00	0.78	Upper
53	4.02	6,192	168	1.00	1.00	0.66	0.89	Upper

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STATEMENT OF CONTRIBUTION TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Jeerasak Chobtang

Name/Title of Principal Supervisor: Sarah McLaren

Name of Published Research Output and full reference:

Chobtang, J., Ledgard, S.F., McLaren, S.J. and Donaghy, D.J. 2016. Life cycle environmental impacts of high and low intensification pasture-based milk production systems: a case study of the Waikato region, New Zealand. Journal of Cleaner Production (accepted pending revision)

In which Chapter is the Published Work: Chapter 4

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate:

  and / or

- Describe the contribution that the candidate has made to the Published Work:

  The candidate was responsible for developing criteria and methods to identify intensification index for dairy farming systems, performing statistical analysis, interpreting the results and writing the majority of the manuscript.

Jeerasak Chobtang — 12 May 2016

Candidate's Signature — **Date**

Sarah McLaren — 16 May 2016

Principal Supervisor's signature — **Date**

Chapter 5 Consequential Life Cycle Assessment of Pasture-Based Milk Production Systems in the Waikato Region, New Zealand

This chapter has been modified and accepted for publication in Journal of Industrial Ecology.

Abstract

Farm intensification options in pasture-based dairy systems are generally associated with increased stocking rates coupled with the increased use of off-farm inputs to support the additional feed demand of animals. However, as well as increasing milk production per hectare, intensification can also exacerbate adverse impacts on the environment. The objective of the present study was to investigate environmental trade-offs associated with potential intensification methods for pasture-based dairy farming systems in the Waikato region, New Zealand. The intensification scenarios selected were (i) increased pasture utilization efficiency (PUE scenario), (ii) increased use of nitrogen (N) fertiliser to boost on-farm pasture production (N fertiliser scenario) and (iii) increased use of brought-in feed as maize silage (MS) (MS scenario). Twelve impact categories were assessed. The PUE scenario was the environmentally preferred intensification method, and the preferred choice between the N fertiliser and MS scenarios depended upon trade-offs between different environmental impacts. Sensitivity analysis was carried out to test the effects of choice associated with: (i) the approaches used to account for indirect land use change (ILUC), and (ii) the competing product systems (conventional beef systems) used to handle the co-product dairy meat for the Climate Change (CC) indicator. Results showed that the magnitude of the CC indicator results was influenced by the ILUC-accounting approaches and the choice associated with a global marginal beef mix, but the relative CC indicator results for the three intensification scenarios remained unchanged.

Keywords: Environmental management ● Dairy ● Farm intensification ● Life Cycle Assessment (LCA) ● Land use ● System substitution

5.1 Introduction

A substantial increase in global demand for food products, including dairy products, over the next decade is anticipated, resulting mainly from increasing world population and greater individual household incomes (Tilman et al. 2011). Additionally, globalization of feed and food systems with the concomitant geographical separation of locations of feed/food production and consumption is likely to lead to increasingly unbalanced nutrient flows, and potentially increased environmental impacts (Bouwman et al. 2013; Lassaletta et al. 2014; Adrian et al. 2015).

New Zealand is the single largest dairy exporting country in the world, accounting for approximately one-third of globally traded dairy products in 2012 (OECD/FAO 2013). Recently, the New Zealand government set a target of doubling the income from agricultural exports, including dairy products by 2025 (New Zealand Government 2012). For the New Zealand dairy sector, this increase can be achieved through a combination of increased total milk production and increased economic value of dairy products (New Zealand Government 2012). In recent years, farms have been intensifying mainly through increased stocking rates, coupled with the use of off-farm inputs (MacLeod and Moller 2006; Pinares-Patiño et al. 2009), leading to a substantial increase in the rate of total milk production compared with the rate of dairy farmland expansion (DairyNZ 2014a). Nevertheless, these ongoing system changes have not been assessed with regard to their comprehensive life cycle impacts on the environment.

Life Cycle Assessment (LCA) is an analytical approach used for assessing multiple environmental impacts over the life cycle of a product (International Organization for Standardization 2006a, 2006b). In principle, there are two modelling approaches in LCA: attributional (ALCA) and consequential (CLCA) (Finnveden et al. 2009; Weidema et al. 2009). While ALCA attempts to quantify environmental impacts and identify environmental hotspots of a product system based on a status quo situation, CLCA accounts for environmental impacts associated with a change in demand for a product (Finnveden et al. 2009; Weidema et al. 2009).

To our knowledge, only three published CLCA studies have assessed environmental consequences of milk production. Thomassen et al. (2008) assessed changes in three environmental impacts (climate change, acidification, eutrophication) arising from an increased demand for milk production in a Dutch confinement-based dairy system (by assuming one more farm was established). Nguyen et al. (2013) assessed a change in climate change impact arising from a dairy system change from maize silage-based to grass-based system in France. Dalgaard et al. (2014) assessed the carbon footprint of milk from an average Danish dairy farming system resulting from an increase in demand for milk. Therefore, the objective of the present study was to investigate the comprehensive environmental trade-offs associated with potential intensification methods to support increased pasture-based milk production in the Waikato region of New Zealand using CLCA.

5.2 Methods

5.2.1 Goal and Scope Definition

The CLCA modelling approach used was generally compliant with the LCA standards (ISO 14040 and 14044), as provided by the International Organization for Standardization (2006a, 2006b). In addition, the generic framework and guidelines for CLCA described by Ekvall and Weidema (2004) and Weidema et al. (2009) were adopted. Material flows were constructed using SimaPro v8 software (Pré Consultants 2013).

The desired milk production was targeted at 20% above the average milk production in the Waikato region in 2010/11 of 12,851 kg fat- and protein-corrected milk (FPCM) per hectare (see Chapter 3 for more details).

5.2.2.1 Selection of intensification methods

The present study focused on increased milk production using an increased stocking rate that was facilitated by different intensification methods. Farm intensification requires either additional off-farm inputs (e.g. brought-in feeds and fertilisers) (MacLeod and Moller 2006; Pinares-Patiño et al. 2009) or increased efficiency in farm management practices (e.g. improved pasture utilization and using new technologies such as robotic milking systems) (Clark et al. 2015).

The selection of farm intensification methods in the present study was based on a generic framework for increased pasture-based milk production in New Zealand (Figure 5.1), showing that the two strategies (increased animal productivity and increased stocking rate) have several consequences which are associated mainly with the provision of feed. Even though other farm attributes (e.g. farm infrastructure and animal medicines) can also be affected by increased levels of farm intensification, the contribution of these farm attributes were not accounted for in the present study due to a lack of inventory information and assumed insignificant contribution (Basset-Mens et al. 2009; O'Brien et al. 2012).

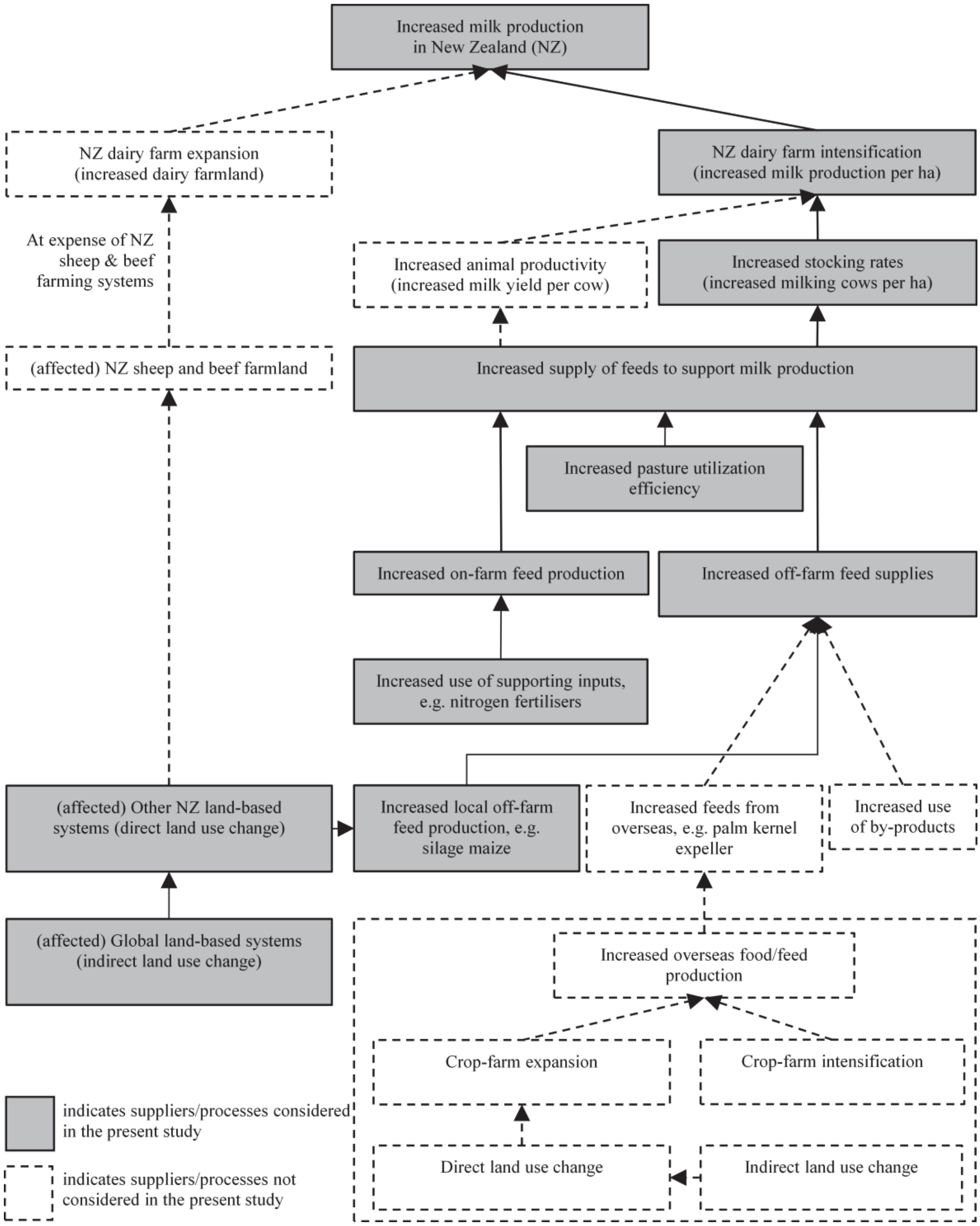

Figure 5.1 A generic framework illustrating the main pathways to increase milk production in New Zealand pasture-based dairy systems; boxes and arrows with solid lines represent the pathways assessed in the present study.

Feed supplies can be increased by a variety of methods; for example, extra feed can be supplied through improved pasture utilization efficiency (PUE). This refers to animals harvesting more feed from existing pastures (i.e. reducing unutilized pasture yield) (McCarthy et al. 2014). Based on data associated with estimated pasture intake (see

Chapter 3) and potential pasture production (Lee et al. 2012) in New Zealand dairy farming systems, there is potential to achieve dairy intensification through improved PUE (McCarthy et al. 2014). In addition, increased use of fertilisers, especially nitrogen (N) fertilisers, can boost pasture production (Ledgard et al. 1999; Vogeler and Cichota 2015). Alternatively, increased demand for feeds can simply be met by increased use of brought-in feeds (e.g. palm kernel expeller [PKE] and maize silage [MS]) (Clark et al. 2013).

Therefore, in the present study, three potential intensification methods were evaluated: (i) increased PUE (PUE scenario), (ii) increased use of N fertiliser as urea (N fertiliser scenario), and (iii) increased use of brought-in feed as maize silage (MS scenario).

5.2.1.2 Scope of the study

This study accounted for environmental impacts associated with extra elementary flows that were actually affected by each of the selected intensification scenarios using a cradle-to-farm gate perspective as a system boundary. In addition to milk, the pasture-based dairy systems generated two other co-products: (i) dairy meat (derived from slaughtered cull cows and surplus calves), and (ii) young stock (live surplus dairy calves) which are reared for meat production (Figure 5.2); the estimated proportions of slaughtered and reared surplus calves were based on Flysjö et al. (2011).

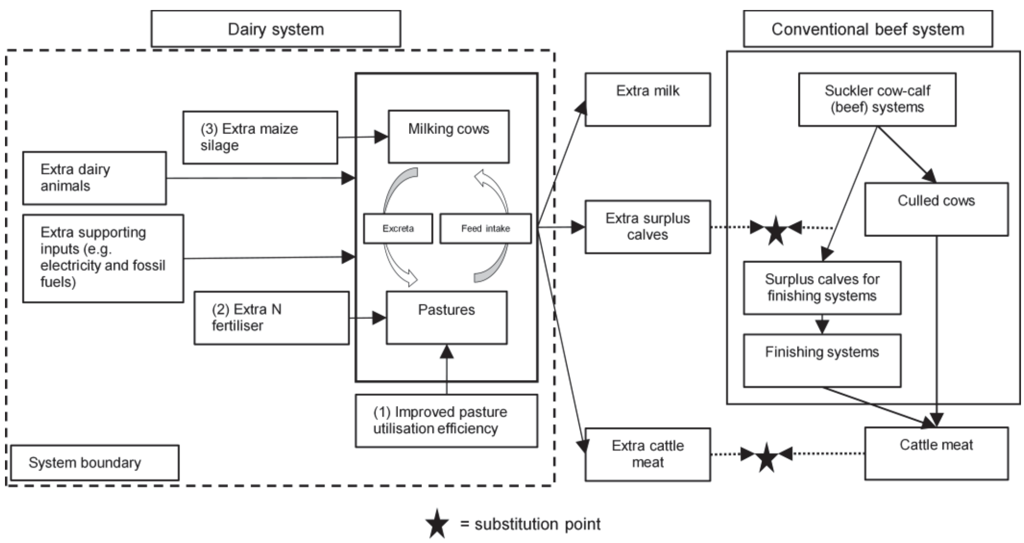

Figure 5.2 Simplified elementary flows and system boundary associated with the use of different intensification methods to increase pasture-based milk production: (1) increased pasture utilization efficiency, (2) increased use of nitrogen (N) fertilisers, and

(3) increased use of brought-in maize silage. Substitution points indicate where the extra dairy co-products displace the products derived from beef farming systems.

5.2.1.3 Functional unit

The functional unit used in the present study was one kg of FPCM. The FPCM was calculated according to the International Dairy Federation (2010): FPCM (kg) = milk yield (kg) × [(0.1226 × fat%) + (0.0776 × protein%) + 0.2534].

5.2.1.4 Co-product handling methods

In CLCA, it is compulsory to avoid allocation by using system expansion and/or system substitution (Weidema and Schmidt 2010; Pelletier et al. 2015). For system substitution, the environmental burdens of co-products were displaced by the burdens of appropriate competing products. Therefore, the remaining burdens were attributed to the studied systems (Ahlgren et al. 2015; Pelletier et al. 2015). Ekvall and Weidema (2004) recommended identifying a competing product system based on its functions and market segments.

In the present study, the dairy co-products (dairy meat and young stock) were assumed to displace the same products from conventional beef systems (Figure 5.2). The substitution rate was 1:1, on a mass basis.

5.2.2 Life Cycle Inventory Analysis

5.2.2.1 Background processes

Elementary flows associated with the background processes are presented in Table D1 of the Supplementary material D. These flows were derived from the ecoinvent v3 database (ecoinvent Centre 2013).

5.2.2.2 Foreground processes

Table 5.1 presents a summary of the quantified input-output flows for each intensification scenario. The equations and emission factors used to calculate the foreground processes are presented in Table D2 of the Supplementary material D.

Table 5.1 Summary of annual elementary flows associated with different intensification scenarios of increased pasture-based milk production in the Waikato region, New Zealand. In all cases, a per-ha basis refers to the on-farm stage only (i.e. area used for grazing and milking dairy cows).

Scenario [a]	Extra milk [b]	Extra inputs				Extra outputs (head/ha) [c]		
		New cow	Urea N	Grazed grass	Maize silage	Cull cow	Surplus calf	Surplus live calf
	(kg FPCM/ha)	(head/ha)	(kg N/ha)	(kg DM/ha)	(kg DM/ha)			
PUE	2,570	0.543	-	2,275	-	0.125	0.191	0.133
N	2,570	0.543	137	2,275	-	0.125	0.191	0.133
MS	2,570	0.543	-	-	2,275	0.125	0.191	0.133

[a] Farm intensification scenarios: PUE = increased pasture utilization efficiency; N = increased use of nitrogen fertiliser; MS = increased use of brought-in maize silage; [b] An increased milk production target was calculated as 20% over milk production per hectare in the Waikato region for the production year 2010/11 of 12,851 kg fat- and protein-corrected milk (FPCM) per ha (the reference system) (see Chapter 3); it was assumed that milk production per cow was the same as that in the reference system: 4,730 kg FPCM per cow.

5.2.2.3 Identification of marginal suppliers

The method to identify the marginal suppliers was based mainly on Weidema (2003) and Ekvall and Weidema (2004). Other modifications are described in the following sections.

As the present study focused on farm intensification through increased stocking rate, extra (new) cows had to be added to the existing dairy system (the reference scenario), and these extra cows were assumed to be derived from replacement heifers. In addition, marginal suppliers were identified for fossil energy and electricity for all intensification scenarios, for N fertiliser and lime for the N fertiliser scenario, and MS for the MS scenario.

5.2.2.4 Rearing of new cows/replacement heifers

Elementary flows in the production of new cows were assumed to be similar to those previously modelled for the rearing of replacement animals, which included the production of maize grain and barley grain (the proxy for concentrate feed at a ratio of 60:40) (Wales et al. 2009), pasture production and other relevant inputs (see Chapter 3).

Elementary flows associated with the marginal production of maize grain and barley grain are presented in Table D3 of the Supplementary material D. The flows and inventory methods for these production systems are presented in Tables D1 and D2, respectively. Environmental impacts associated with the marginal production systems of maize grain and barley grain, and the rearing of replacement animals are presented in Table D4 of the Supplementary material D.

5.2.2.5 Marginal fossil energy and electricity

Data associated with demand for fossil fuels (diesel and gasoline) and electricity for the milk production system in New Zealand were derived from StatisticsNZ (2014) (see Table D5 of the Supplementary material D). The consequential flows of diesel and gasoline were derived from the ecoinvent v3 database (ecoinvent Centre 2013). The New Zealand marginal electricity grid mix was identified according to Treyer and Bauer (2014), comprising 73% hydropower, 20% geothermal power and 7% wind power (see Table D1 for more details). The share of these electricity sources has increased over the past ten years and is expected to increase in the future (with the encouragement of the New Zealand government); the share of fossil-based electricity (e.g. coal-fired, natural gas and oil) has decreased over the same period (Ministry of Business Innovation and Employment 2013). These unconstrained electricity sources were then combined proportionally (Ministry of Business Innovation and Employment 2013) to form a marginal electricity grid mix using corresponding consequential electricity flows (rest-of-the world) derived from ecoinvent database (ecoinvent Centre 2013).

5.2.2.6 Scenario-specific marginal suppliers

Specific marginal suppliers refer to the suppliers that were actually affected by the decision to increase milk production for each intensification scenario.

The PUE scenario Apart from the common marginal suppliers mentioned above, there were no specific marginal suppliers associated with this intensification scenario.

The N fertiliser scenario Urea is the major form of N fertiliser used across the world and its use has increased over the past decades (FAO 2015). In New Zealand, urea accounted for about 90% of total N fertiliser use, of which about 70% was imported from overseas (StatisticsNZ 2015). Therefore, in the present study, urea was identified as the marginal fertiliser, and assumed to be derived from a global market. To estimate the amounts of N fertiliser (urea) required to boost pasture production, the response rate of pasture growth to N fertiliser applied in the Waikato region was used, according to Li et al. (2011), as presented in Table D6 of the Supplementary material D. Sinclair et al. (1993) recommended using 0.5 kg of lime per kg of urea N to neutralize the potential soil acidification effect. It should be noted that the typical pasture N concentration in dairy pastures in New Zealand (3.7% N, on a dry matter basis) was used for pasture N concentration.

The MS scenario Two main brought-in feeds are commonly used in New Zealand pasture-based dairy systems: PKE and MS. These two feed sources accounted for 60% and 33% of the total brought-in feeds used in dairy farming systems in the Waikato region for the production year 2010/11, respectively (data derived from DairyBase: http://www.dairynz.co.nz). However, it is unlikely that PKE could promptly respond to an increased demand for brought-in feeds since it is a minor co-product of the oil palm industry (Schmidt 2015). Therefore, MS was identified as the only marginal supplier and assumed to be the only brought-in feed used in this intensification method. It was assumed that MS production per hectare in New Zealand was at its maximum capacity. As a result, increased demand for MS was derived from the maize farming system displacing other arable crop systems. Elementary flows of MS production are presented in Table D3. The proxy flows and inventory methods used for MS production are included in Tables D1 and D2, respectively. Multiple environmental impacts of the MS production system are presented in Table D4. In determining the total amount of brought-in MS, the actual amount of feed offered was considered feed waste during feeding-out, as recommended by DairyNZ (2015).

5.2.2.7 Identification of competing product systems

Identification of competing product systems for use to handle the corresponding co-products was based mainly on Ekvall and Weidema (2004) who recommended that the competing product systems should have the same function or market segmentation as the corresponding co-product.

In the present study, beef products derived from conventional beef systems in New Zealand were identified to be competing product systems (see Figure 5.2). Note that conventional beef production systems in New Zealand are export-oriented (Morris and Kenyon 2014) and the total amount of New Zealand exported beef products increased over the past decade (FAO 2015). Additionally, New Zealand is the fourth largest beef exporter in the world, after Brazil, Australia and the USA (FAO 2015). The associated elementary flows and their multiple environmental impacts for the New Zealand beef farm systems are presented in Tables D7 and D8 of the Supplementary D, respectively.

Alternatively, beef marketing can be considered to occur at a global level (Dalgaard et al. 2014); however, an environmental inventory for a global mix of the beef systems actually affected has not yet been estimated. Therefore, in the present study, the global beef mix was established and tested in sensitivity analysis (described in Section of Sensitivity Analysis).

5.2.3 Life Cycle Impact Assessment

Twelve midpoint impact categories were assessed: Climate Change (CC), Ozone Depletion Potential (ODP), Human Health Toxicity - non-cancer effects (Non-cancer), Human Health Toxicity - cancer effects (Cancer), Particulate Matter (PM), Ionizing Radiation - human health effects (IR), Photochemical Ozone Formation Potential (POFP), Acidification Potential (AP), Terrestrial Eutrophication Potential (TEP), Freshwater Eutrophication Potential (FEP), Marine Eutrophication Potential (MEP) and Ecotoxicity for Aquatic Freshwater (Ecotox), as recommended by the European Food SCP Roundtable (2013) for assessing environmental performance of food and drinks products.

5.2.4 Sensitivity Analysis

Sensitivity analysis was performed to investigate the influence of (i) different approaches to account for the contribution of ILUC, (ii) the choice associated with which competing product systems (beef systems) to use, and (iii) the choice associated with different characterization models. It should be noted that the sensitivity analysis was focused only on the CC impact indicator.

For the analysis associated with ILUC, the increased demand for off-farm inputs (e.g. feed crops and new cows) was linked to the displacement of other land-based systems. It was assumed that the production of new cows and MS displaced existing land-based agricultural systems in New Zealand, and their impacts on the CC indicator were assumed to be minimal and not accounted for (i.e. no direct LUC).

However, the displaced land-based systems resulted in reduced corresponding products delivered. Assuming that the demand for the displaced products was constant, these displaced products needed to be produced somewhere else, and eventually will have resulted in conversion (land transformation) of new land (e.g. natural grassland or forest), leading to ILUC. Table 5.2 presents the extra demand for land, land saved as a result of the extra dairy co-products (avoided land use) and the net area of ILUC for each intensification method, assuming that a hectare of displaced land resulted in conversion of a hectare of natural grassland or forest.

Table 5.2 A summary of extra demand for agricultural land, land saved as a result of dairy co-product displacement and net agricultural land required (m^2 per kg extra fat- and protein-corrected milk).

Land class	Intensification scenarios [e]		
	PUE	N	MS
Pasture [a]	0.56	0.56	0.56
Cropland [b]	0.17	0.17	0.56
Avoided pasture [c]	-1.02	-1.02	-1.02
Net (total) [d]	-0.29	-0.29	0.10

[a] Associated with the production of new cows and extra replacement animals; [b] Associated with production of maize silage and feed grains; [c] Associated with land saved as a result of the displacement of the dairy co-products of dairy meat and live surplus calves to conventional beef systems; [d] Assuming that the displacement ratio for extra demand for any agricultural land (pasture and crop lands) and natural land (e.g. natural grassland and forest) was 1:1; [e] PUE = increased pasture utilization efficiency; N = increased use of nitrogen fertilisers; MS = increased use of brought-in maize silage.

Two approaches to account for the effects of ILUC were tested. For the first, the total amount of natural land to be converted was calculated using a cause-effect relationship that accounted for productivity of converted land, according to Schmidt et al. (2015) (see Table D9 of the Supplementary material D for more details). For the second, emission factors to assess ILUC impacts resulting from the expansion of arable land, at the expense of deforestation for individual regions (biomes) across the world, based on Tonini et al. (2015) were used (see Table D9).

For the analysis associated with the choice of competing beef systems, the weight-averaged CC impact indicator derived from the world's top three beef exporting countries (FAO 2015) was considered to represent the global beef system affected by increased co-product of dairy meat (see Table 5.3). Beef production and exports in these countries have increased over the past ten years (FAO 2015). Note that the CC indicator results of these competing beef systems were derived from literature (see Table 5.3) where ALCA was used and the co-products were handled using allocation methods. Therefore, it is recommended that the results are used for a theoretical comparison only.

Table 5.3 Relative market share of exported beef in the world's top three beef exporting countries that were used to calculate a global (weighted-average) marginal beef mix and their Climate Change (CC) indicator results. Note that the CC results exclude beef derived from dairy systems and land use change.

Exporting countries	Relative market share (%) [a]	Climate Change (kg CO_2 eq. per kg live weight)	Reference
Brazil	37	17.6	Cederberg et al. (2011)
Australia	37	10.8	Ridoutt et al. (2011)
USA [b]	26	14.8	Pelletier et al. (2010)
Weighted-average		14.4	

[a] Recent market information for total beef exports in the world's top three beef exporting countries was based on FAO (2015), collectively accounting for ~51% of total internationally traded beef products (meat without bone); [b] Data were based on a feedlot-finished beef system.

For the analysis associated with the choice of characterization models, recently-updated characterization factors for greenhouse gases for the CC impact indicator for a 100-year time horizon based on Myhre et al. (2013) were tested and compared with the default method based on the Intergovernmental Panel on Climate Change (2007).

5.3 Results

5.3.1 Environmental Impacts

The results for 12 environmental impact indicators associated with the three intensification scenarios are depicted in Figure 5.3. The PUE scenario had the lowest results for nine of the indicators. The exceptions were the IR indicator, which was higher than that in the N fertiliser scenario (but lower than that in the MS scenario), and the AP and TEP indicators, which were slightly higher than those in the MS scenario (but lower than those in the N fertiliser scenario).

For the MS scenario, results for seven indicators (ODP, Non-cancer, Cancer, IR, POFP, FEP and Ecotox) were higher than those for the N fertiliser scenario.

For the N fertiliser scenario, results for five indicators (CC, PM, AP, TEP and MEP) were higher than those for the MS scenario. Net values of indicators were partly influenced by the contribution of the displaced beef systems, especially those that were relatively high for the beef systems such as CC, Non-cancer, Cancer, POFP, FEP and MEP.

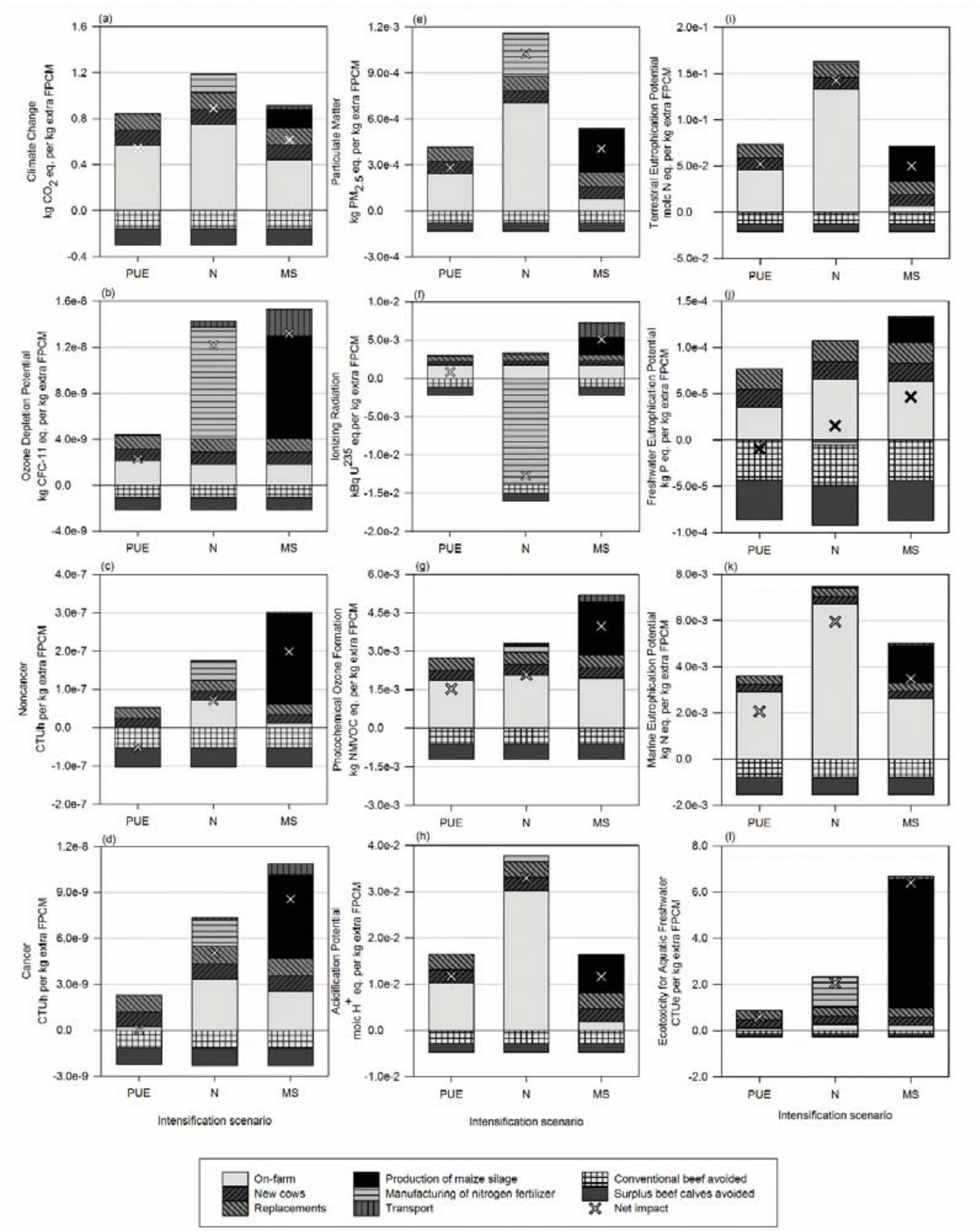

Figure 5.3 Environmental indicators and the contribution of life cycle stages associated with increased pasture-based milk production in the Waikato region, New Zealand, through the use of different intensification methods: (a) Climate Change, (b) Ozone Depletion Potential, (c) Human Health Toxicity (non-cancer effects), (d) Human Health

Toxicity (cancer effects), (e) Particulate Matter, (f) Ionizing Radiation (human health effects), (g) Photochemical Ozone Formation Potential, (h) Acidification Potential, (i) Terrestrial Eutrophication Potential, (j) Freshwater Eutrophication Potential, (k) Marine Eutrophication Potential and (l) Ecotoxicity for Aquatic Freshwater. Farm intensification scenarios: PUE = increased pasture utilization efficiency; N = increased use of nitrogen fertiliser; MS = increased use of brought-in maize silage.

5.3.2 Results of sensitivity analysis

5.3.2.1 Choice associated with ILUC accounting methods

The two ILUC-accounting approaches (Schmidt et al. 2015; Tonini et al. 2015) were chosen because they represented different ILUC-accounting perspectives. Schmidt et al. (2015) provided a conceptual framework for ILUC by linking a change in demand for particular land-based products in one region with its consequential impact(s) in other regions on a per-hectare basis, using a cause-effect chain (a biophysical model). In contrast, Tonini et al. (2015) based their ILUC-accounting approach on global land-conversion (accounting for both carbon and nitrogen cycles) and land use intensification (emissions associated with production and use of chemical fertilisers), and then attributed the impacts to an increase in demand for one hectare of arable land, using a deterministic model.

Inclusion of ILUC-emissions changed the magnitude of the CC indicator results, depending on the accounting approach (Figure 5.4). Using the approach of Schmidt et al. (2015) increased the net CC indicator results for the PUE, N fertiliser and MS scenarios by 1%, 1% and 14%, respectively. In contrast, using the approach of Tonini et al. (2015) decreased the net CC indicator results for the PUE and N fertiliser scenarios by 22% and 13%, respectively, but increased it for the MS scenario by 7%. However, even though different ILUC accounting approaches resulted in changes in magnitude of the CC indicator results, they did not alter the relative ranking of the results for the intensification scenarios.

In summary, the impacts associated with ILUC can be large and affected the LCA results of dairy farming systems, especially when different land-related intensification methods are involved (Flysjö et al. 2012; Nguyen et al. 2013). In the present study, avoided impacts resulting from avoided land use were apparent when the intensification methods involved increased feed supply within a dairy farm (i.e. the PUE and N fertiliser scenarios), whereas impacts resulting from increased arable land to produce additional maize silage in the MS scenario led to extra impacts resulting from ILUC.

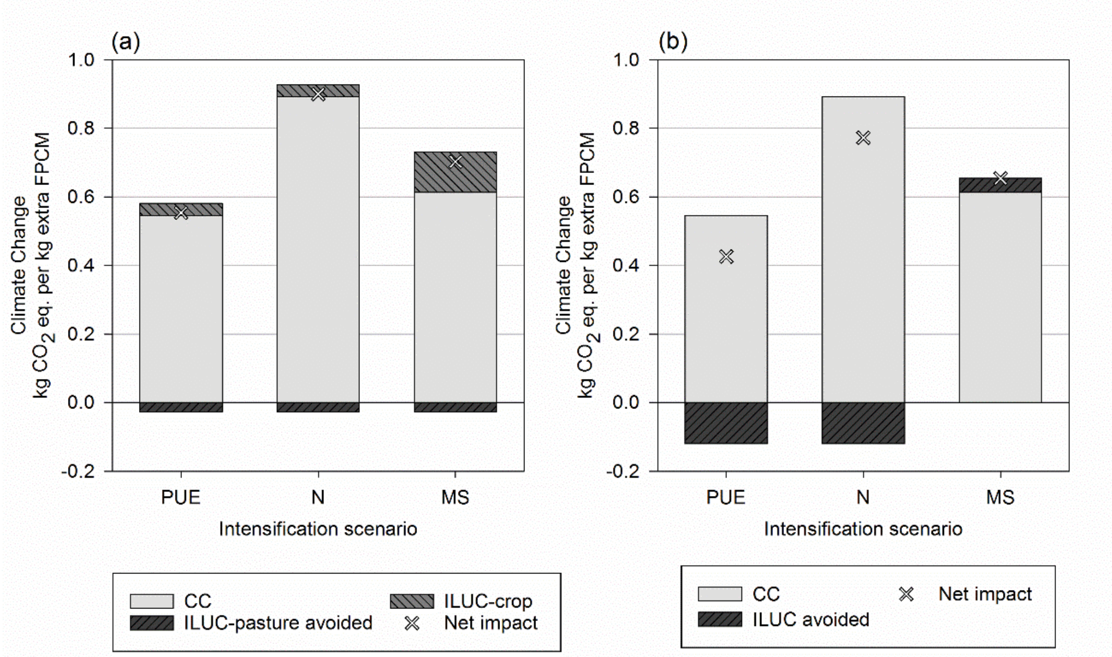

Figure 5.4 The effects of choice associated with indirect land use change (ILUC) accounting approaches on Climate Change (CC) indicator results: (a) the accounting approach according to Schmidt et al. (2015), for which ILUC-pasture refers to the contribution of ILUC associated with pastures in dairy and beef systems, and ILUC-crop refers to the contribution of ILUC associated with feed-crop production systems; (b) the accounting approach according to Tonini et al. (2015). Farm intensification scenarios: PUE = increased pasture utilization efficiency; N = increased use of nitrogen fertiliser; MS = increased use of brought-in maize silage.

5.3.2.2 Choice associated with competing beef systems

The choice associated with displaced beef systems contributed significantly to the CC indicator result of the three intensification scenarios (Figure 5.5a). Net CC indicator results (per kg extra FPCM) decreased by 40% (from 0.55 kg CO_2 eq. for the default New Zealand marginal beef system to 0.33 kg CO_2 eq. for the global marginal beef mix) for the PUE scenario, by 24% (from 0.89 to 0.68 kg CO_2 eq.) for the N fertiliser scenario, and by 35% (from 0.61 to 0.40 kg CO_2 eq.) for the MS scenario (Figure 5.5a). Even though the different displaced beef systems changed the magnitude of the CC indicator results, they did not alter the relative ranking of the results for the intensification scenarios. This was because all the intensification scenarios had similar quantities of the co-product of dairy meat (see Table 5.1).

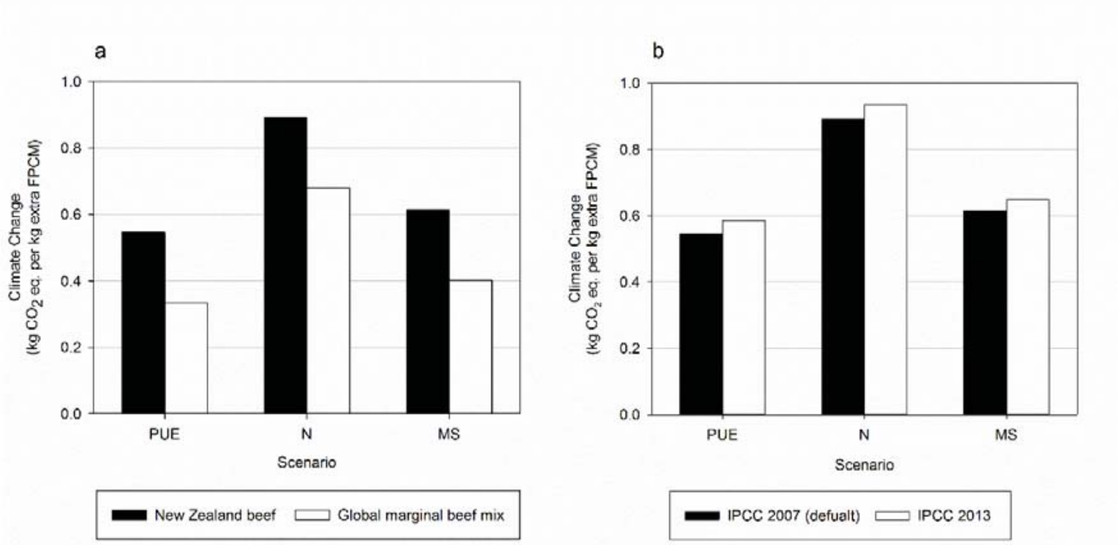

Figure 5.5 The effects of choice associated with: (a) displaced beef systems that were used to handle the extra co-product of dairy meat for different intensification scenarios and (b) different characterization models (IPCC 2007: characterization factors based on the Intergovernmental Panel on Climate Change (2007) versus IPCC 2013: characterization factors based on Myhre et al. (2013)) on net Climate Change indicator results. Farm intensification scenarios: PUE = increased pasture utilization efficiency; N = increased use of nitrogen fertiliser; MS = increased use of brought-in maize silage.

5.3.2.3 Choice associated with characterisation models

The net CC indicator results derived from the IPCC 2013 characterization factors for all intensification methods increased (by 7%, 5% and 6% for the PUE, N fertiliser and MS scenarios, respectively) compared with those derived from the IPCC 2007 (Figure 5.5b). However, the relative ranking of the intensification options did not change.

5.4 Discussion

This study primarily aimed to assess environmental impacts associated with increased milk production through the use of different intensification options using CLCA. As such, it accounted only for environmental impacts associated with additional marginal activities and competing (displaced) product systems (i.e. meat and live surplus beef calves derived from conventional beef systems). The marginal suppliers and competing product systems were identified using the stepwise method of Ekvall and Weidema

(2004). This identification method has successfully been used in several CLCA studies on different product systems (e.g. soybean meal (Dalgaard et al. 2008), cow's milk (Thomassen et al. 2008) and biodiesel (Reinhard and Zah 2011)).

5.4.1 Environmental trade-offs

The main advantage of LCA compared with other environmental management tools is its ability to account for comprehensive environmental emissions and prevent burden-shifting between life cycle stages, impact categories or interrelated product systems (Finnveden et al. 2009; Hellweg and Milà i Canals 2014). However, interpretation of LCA results presented as midpoint environmental indicators (multiple environmental indicators) can be difficult, especially when a management option results in a reduction in one environmental indicator but an increase in one or more other environmental indicators (Page et al. 2012; Jolliet et al. 2014).

The present study highlighted that no one intensification scenario completely outperformed the other scenarios for all indicators assessed.

5.4.1.1 PUE scenario

It has been reported that maximizing utilization of pasture is one of the best farm practices for pasture-based dairy farming systems, potentially resulting not only in improved farm economic returns but also in a reduction in a range of environmental impacts per kg product (Cederberg and Stadig 2003; O'Brien et al. 2015).

The PUE scenario can be considered as the most advantageous. For this scenario, the total demands for extra inputs for this intensification option were fewer than those for the two other intensification options, since extra feed supply was derived solely from existing pastures. This was the main reason why it had lower impacts than the two other options, except for IR, AP and TEP. The higher IR impact for the PUE scenario than for the N fertiliser scenario was due to an assumption associated with imported N fertiliser. Manufacture of N fertilisers produces heat that, upon utilization, displaces household heat production derived from electricity generated by nuclear power plants in European countries (ecoinvent Centre 2013). In addition, the higher AP and TEP impacts for the PUE scenario than for the MS scenario were associated with the higher N concentrations of pastures compared with MS, leading to higher surplus N intake, and subsequently resulting in higher N excreted, especially urinary N (Ledgard et al. 2009;

Selbie et al. 2015). These urinary N-based emissions, especially ammonia (NH_3), subsequently contributed to an increase in AP and TEP indicator results.

In summary, increased PUE can be a promising method to increase pasture-based milk production while minimizing most environmental impact categories. However, increased milk production through this intensification option will be limited by the productive capacity of pastures (McCarthy et al. 2014).

5.4.1.2 N fertiliser scenario

For the N fertiliser scenario, apart from N emissions occurring during the manufacturing processes (upstream process) of N fertilisers (Hasler et al. 2015), the use of N fertilisers (downstream process) released several emissions, such as CO_2, NH_3, nitrous oxide (N_2O), nitrogen oxides (NO_x) and nitrate (NO_3^-) (Ledgard et al. 2009; Nemecek et al. 2014), that contributed to a range of environmental impacts.

For this intensification option, the impacts of CC, Non-cancer, Cancer, PM and Ecotox were increased by the emissions generated from the manufacture of N fertilisers. The emissions produced from the on-farm stage were associated with the use of N fertilisers (NH_3, N_2O and NO_3^-), feed digestion (enteric CH_4) and animal excreta (NH_3, N_2O, NO_3^- and VOCs).

It is important to note that the increased use of N fertilisers can result in increased pasture N concentration (Ledgard et al. 1999; Vogeler and Cichota 2015), resulting in surplus N intake by animals. This surplus N ingested is subsequently excreted mainly through urination and is prone to be lost through different environmental compartments, e.g. NH_3 to atmosphere and NO_3^- to surface water and groundwater (Ledgard et al. 2009; Monaghan and de Klein 2014) and contributes to a range of environmental impacts (e.g. PM, AP, TEP and MEP). However, this aspect was not captured in the present study due to a lack of appropriate inventory models.

In summary, dairy intensification through the use of N fertilisers is associated largely with increased environmental impacts driven by N-based emissions. Maximizing N use efficiency is likely to be one of the most promising mitigation options to minimize such environmental impacts. It is important to note that yield response to N fertiliser by pastures decreases at high N inputs and limits the potential of this intensification option.

5.4.1.3 MS scenario

Extra inputs were required to produce the extra MS for use in the MS scenario; these generated additional environmental emissions that contributed to a range of indicators. For example, use of P fertilisers released heavy metals to ecosystems, causing increased Non-cancer, Cancer and Ecotox results, as these indicators were predominantly driven by heavy metals (EC-JRC-IES 2012; Yang 2013).

The MS scenario reduced N_2O, NH_3 and NO_3^- emissions at the on-farm stage (due to lower N concentration in MS), which resulted in a reduction in CC, PM, AP, TEP and MEP results compared with the N fertiliser scenario. In contrast, this intensification option was associated largely with additional releases of NO_x (mainly from extra transport), pesticides and related emissions (e.g. ozone depleting substances), heavy metals and P fertiliser-related emissions and P runoff, resulting in higher net impacts on ODP, Non-cancer, Cancer, IR, POFP, FEP and Ecotox than for the N fertiliser scenario.

In summary, even though MS is increasingly becoming one of the most promising brought-in feed supplements in dairy systems in New Zealand (Pinares-Patiño et al. 2009; Ledgard and Boyes 2013), the above-mentioned environmental trade-offs must carefully be considered and addressed accordingly.

5.4.2 Methodological issues

In CLCA, uncertainties about the choice of marginal suppliers and competing product systems can reduce confidence in the reliability of the results, and identification of these choices is not straightforward (Ekvall and Weidema 2004; Brandão et al. 2014). Future development of accounting approaches is required to quantify impacts associated with ILUC, and there is no consensus method (Warner et al. 2014; De Rosa et al. 2016), even though some economic modelling approaches can be used (Kløverpris et al. 2008; Earles and Halog 2011).

5.4.2.1 Choice of ILUC accounting methods

Studies have found that the contribution of ILUC to the CC indicator for land-based product systems varies depending on the accounting methods (Dumortier et al. 2011; Panichelli and Gnansounou 2015) and their underlying assumptions (Kim et al. 2014; Ahlgren et al. 2015).

Results of the sensitivity analysis highlighted that different ILUC accounting approaches changed the magnitude of the net CC results but did not affect the relative ranking of the intensification scenarios. This change was due to differences in extra land demands between the intensification scenarios.

In summary, different magnitudes of net CC impacts were obtained from using different ILUC accounting approaches. Depending on ILUC accounting approaches, some intensification scenarios resulted in a reduction in the CC indicators, for example, for the PUE and N fertiliser scenarios when the approach of Tonini et al. (2015) was used. However, the contribution of ILUC increased net CC impacts for the MS scenario regardless of accounting approaches, due to extra emissions associated with the extra demand for land used to produce additional MS.

5.4.2.2 Choice of competing product systems

Earles and Halog (2011) reported that identification of competing product systems in CLCA can be difficult since the actual displacement resulting from the increase in co-product of the analysed systems is usually not straightforward. Additionally, it has been recommended that economic modelling approaches be used to identify competing product systems and quantify the size of displacement (Kløverpris et al. 2008; Earles and Halog 2011). Chalmers et al. (2015) reported that it may be incorrect to use a 1:1 substitution ratio between two functionally equivalent product systems, and recommended using an approach that can account for substitution effects associated with marketing (e.g. market value) to quantify potential demand and size of the affected product(s). Nonetheless, a 1:1 substitution ratio (on a mass basis) between the co-products of dairy systems and the displaced beef systems was used in the present study, even though the economic value of these products may differ (Thiesen et al. 2008). Therefore, interpretation of the present results should take this simplification into account.

In summary, identifying the affected product system as a result of an increase in a functionally equivalent product remains challenging and requires a comprehensive modelling approach that can reflect the actual market situation of the displaced product(s). This is because the displaced product identified can have significant effects on the final CLCA results.

5.4.2.3 Choice of characterisation models

Owsianiak et al. (2014) and Bueno et al. (2015) reported that use of different impact assessment methods resulted in different patterns of environmental contribution and subsequently different LCA results, and emphasized the need to perform sensitivity analyses in order to show transparency and support decision-making with the most comprehensive environmental information and make LCA studies more reliable.

In this study, the increase in the CC indicator results as a result of changing from the IPCC 2007 to 2013 characterization factors were associated mainly with the changes in characterization factors for methane (CH_4) (increased from factors of 22 and 25 to 25.25 and 28 for biogenic and fossil CH_4, respectively) and N_2O (decreased from a factor of 298 to 265) (Myhre et al. 2013).

5.5 Conclusions

The PUE scenario was the environmentally preferred intensification option. The preferred choice between the N fertiliser and MS scenarios depends upon trade-offs between environmental impacts. The findings of the present study highlighted that focusing on one or a few impact categories can be misleading in environmental assessment for pasture-based dairy farming systems, since a reduction in one impact category can unintentionally result in an increase in other impact categories.

Decisions can initially be based on the most relevant environmental indicators that have been prioritized and selected, and the selected indicators should reflect scales of decision (i.e. local, regional, national or global) and priority of environmental concern, accordingly. In addition, this present study has clearly shown that the choices associated with ILUC accounting approaches, characterization models, and displaced beef systems significantly changes the magnitude of the CC indicator results, although this change did not alter the relative ranking of the net CC results for different intensification options.

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Supplementary material D

Table D1 A summary of proxy flows used in the consequential LCA of pasture-based dairy farming systems. These flows were derived from ecoinvent Centre (2013).

Materials (flows)	Proxy flows
Seeds/planting materials	Seeds for sowing (Global), market for, consequential
Nitrogen fertilisers	
• Urea	Urea, as N (Global), market for, consequential
• Diammoniun phosphate	Diammonium phosphate prod., as N (Europe), consequential
Phosphorus fertilisers	
• Diammoniun phosphate	Diammonium phosphate prod., as P_2O_5 (Europe), consequential
• Single superphosphate	Single superphosphate prod., as P_2O_5 (Europe), consequential
Pesticides	
• Generic pesticides	Pesticide, unspecified (Global), market for, consequential
• Glyphosate	Glyphosate, unspecified (Global), market for, consequential
Fossil fuels	
• Diesel	Diesel at storage (Europe), market for, consequential
• Petrol	Petrol, low sulphur, (Rest of the world), market for, consequential
• New Zealand electricity grid mix [a]	Marginal electricity grid mix
Domestic transports	
	Lorry >32 metric ton, EURO3 (Global), market for, consequential
	Lorry 7.5-16 metric ton, EURO3 (Global), market for, consequential
	Lorry >32 metric ton, EURO3 (Global), market for, consequential
International transports	Transport, freight, sea, transoceanic ship (Global), market for, consequential

[a] At the time of this study, consequential flow data associated with the electricity grid mix for New Zealand did not exist in the ecoinvent v3 database. Therefore, a marginal electricity grid mix for New Zealand was developed using associated trends in the share of electricity generation technologies in New Zealand, derived from the Ministry of Business Innovation and Employment (2013). Constrained technologies, as recommended by Treyer and Bauer (2014), were excluded. The shares of unconstrained technologies were then calculated as being in the same proportions as the current mix of these technologies in New Zealand: 73% hydro, 20% geothermal and 7% wind power plants. These electricity generation flows were derived from consequential (rest-of-the world) flows in ecoinvent v3 database (ecoinvent Centre 2013).

Table D2 A summary of common inventory methods and emission factors used to quantify environmental emissions in the foreground process of the life cycle of pasture-based milk production

Emissions	Emission factor/models	Units	Sources
Methane (CH_4)			
• Enteric fermentation	0.065 × total GEI [a]/55.65	kg CH_4	MPI (2013)
• Faecal emissions	0.00098198 × on-pasture FDM [b]	kg CH_4	MPI (2013)
• Farm dairy effluent (FDE)	0.06397826 × effluent FDM	kg CH_4	MPI (2013)
Nitrous oxide (N_2O)			
• Urine N deposited onto pastures [c]	0.0116 × urinary N	kg N_2O-N/kg urinary N	Kelliher et al. (2014)
• Faecal N deposited onto pastures [c]	0.0023 × faecal N	kg N_2O-N/kg faecal N	Kelliher et al. (2014)
• Application of FDE	0.01 × FDE N applied	kg N_2O-N/kg effluent N	MPI (2013)
• Nitrogen (N) fertilisers	0.01 × fertiliser N	kg N_2O-N/kg fertiliser N	MPI (2013)
• Leached nitrogen	0.0075 × leached N	kg N_2O-N/kg NO_3-N	MPI (2013)
• Deposition of volatilized NH_3	0.01 × total NH_3-N loss	kg N_2O-N/kg NH_3-N	MPI (2013)
Nitrogen oxides (NO_x)			
• On-farm NO_x emissions	0.21 × N_2O (kg)	kg NO_x	Nemecek and Schnetzer (2011)
• Silage NO_x loss	0.5 × total N-silage gaseous loss	kg NO_x	Del Prado et al. (2011)
Ammonia (NH_3)			
• Animal excreta [d]	0.1 × total on-pasture N excreted	kg NH_3-N/kg N	MPI (2013)
• Farm dairy effluent	0.1 × total effluent N	kg NH_3-N/kg N	MPI (2013)
• Nitrogen fertilisers	0.1 × fertiliser N	kg NH_3-N/kg N	MPI (2013)
• Silage	0.5 × total N-silage loss	kg NH_3-N	Del Prado et al. (2011)
Nitrate (NO_3^-)			
• Leached NO_3^-	Overseer	kg NO_3-N/ha/yr	Wheeler et al. (2003)
Carbon dioxide (CO_2)			
• Urea application	0.773	kg CO_2/kg urea	IPCC (2006)
• Lime application	0.412	kg CO_2/kg lime	IPCC (2006)
• Combustion of diesel	3.12	kg CO_2 eq/kg diesel	Nemecek and Kägi (2007)
• Combustion of petrol	3.00	kg CO_2 eq/kg petrol	Nemecek and Kägi (2007)
Phosphorus (P)			
• P losses	Overseer	kg P/ha/yr	Wheeler et al. (2003)
Volatile Organic Compounds (VOCs)			
• Manure on pasture	28.71	g VOCs/kg DM manure	Rotz et al. (2013)
• Stored manure	60.06	g VOCs/kg DM manure	Rotz et al. (2013)
• Maize silage	34.18	mg VOCs/kg DM silage	Rotz et al. (2013)
• Grass silage	44.07	mg VOCs/kg DM silage	Rotz et al. (2013)
Pesticides [e]	See Table D2.1		
Heavy metals [f]			

[a] GEI = gross energy intake, gross energy contents of feed were estimated using energy factors 23.4 MJ/kg protein, 39.7 MJ/kg fat and 17.4 MJ/kg carbohydrates (Waghorn 2007) where chemical composition of feed was derived from Kolver (2000); [b] FDM = faecal dry matter output, estimated proportion of faecal dry matter deposited onto pastures was based on Ledgard and Brier (2004): ~95% returned directly through grazing and ~5% collected during milking and applied back onto pasture; [c] Estimated proportion of urinary N: faecal N was based on Luo and Kelliher (2010); [d] Estimated

total N in animal excreta was computed using a balance method: N excreta = N intake – N retention (milk, weight gain and foetus) (MPI 2013); [e] Total amounts of pesticides use in different production systems were derived from Manktelow et al. (2005), and release of pesticide active ingredients to different environmental compartments (air, water and soil) were modelled according to the European Environment Agency (EEA) 'Air Pollutant Emission Inventory Guidebook' (Webb et al. 2013); pesticide released into water was based on Kellogg et al. (2002) at 5% of total pesticides applied; the remaining pesticides were assumed to be deposited onto agricultural soils, and capped at 85%, as recommended by Audsley et al. (2003) (see table S2.1); [f] The heavy metal contents in fertilisers were derived from a major fertiliser company in New Zealand (S.F. Ledgard, unpublished); for inventory analysis associated with heavy metals, as there are no widely accepted models to estimate releases of heavy metals to environmental compartments, it was assumed that all heavy metals were deposited onto agricultural soils; this assumption was also applied for the production of feed crops and pastures for the rearing of heifers, and has been used in a number of recent LCA studies (e.g. Astudillo et al. 2014; Niero et al. 2014; Yang 2013); however, it is important to note that this assumption can potentially underestimate toxicity since releases of heavy metals to water generally have higher impacts than those released to agricultural soils (Rosenbaum et al. 2008).

Table D2.1 Factors used to estimate pesticide active ingredients (a.i.) released to the air, water and agricultural soils

Compartment	Vapour pressure (mPa) [a]	Fractions of pesticide a.i. (%)	References
Air	mPa > 10	95	Webb et al. (2013)
	1 < mPa < 10	50	
	0.1 < mPa < 1	15	
	0.01 < mPa < 0.1	5	
	mPa < 0.01	1	
Water		5	Kellogg et al. (2002)
Soil [b]		100 – (air + water)	Audsley et al. (2003)

[a] Vapour pressure of individual pesticides was derived from the PPDB (2014); [b] The remaining pesticides were assumed to be deposited onto soils and capped at 85% (Audsley et al. 2003).

Table D3 Primary inputs used to compute environmental impacts of input products in the off-farm stage (per hectare). These data were derived from industry sources (S. F. Ledgard, unpublished).

Inputs	Units	Maize grain	Barley grain	Maize silage
1. Seeds	kg	90	135	32
2. Chemical fertilisers				
2.1 Nitrogen (N) [a]	kg N	145	98	183
2.2 Phosphorus (P) [b]	kg P	23	15	38
2.3 Lime	kg	500	500	500
3. Pesticides				
3.1 Glyphosate	kg active ingredient	0.91	2.7	0.81
3.2 Pesticides	kg active ingredient	3.89	2.2	3.12
4. Plastic sheet	kg	-	-	8
5. Diesel	kg	111	72	168
6. Electricity	kWh	3.91	3.91	3.91
7. Yields	kg DM (dry matter)	10,235	5,874	23,000

[a] urea was used as a proxy for N fertiliser for all product systems; [b] diammonium phosphate and single superphosphate were used as proxies for P fertiliser for crop and pasture production systems, respectively.

Table D4 Multiple environmental impacts associated with the rearing of replacement heifers and the production of maize grain and barley grain calculated on a per-kg of their functional units (FU) basis. The impacts were based on a cradle-to-farm gate perspective for individual systems.

Impacts	Units	Replacement animal [a] (FU = 1 kg live weight)	Maize system [b] (FU = 1 kg DM maize grain)	Barley system [b] (FU = 1 kg DM barley grain)	Maize silage system [b] (FU = 1 kg DM maize silage)
CC	kg CO_2 eq.	2.03E+01	3.45E-01	3.64E-01	1.43E-01
ODP	kg CFC-11 eq.	1.46E-07	1.80E-08	1.99E-08	7.89E-09
Non-cancer	CTUh	6.80E-06	3.61E-07	4.42E-07	2.08E-07
Cancer	CTUh	1.49E-07	1.09E-08	1.12E-08	4.78E-09
PM	kg $PM_{2.5}$ eq.	1.00E-02	3.67E-04	4.32E-04	2.43E-04
IR	kg U^{235} eq.	1.44E-01	7.05E-03	5.56E-04	2.12E-03
POFP	kg NMVOC eq.	8.04E-02	1.48E-03	1.57E-03	1.84E-03
AP	molc H^+ eq.	3.74E-01	8.64E-03	1.11E-02	7.18E-03
TEP	molc N eq.	1.64E+00	3.64E-02	4.69E-02	3.20E-02
FEP	kg P eq.	5.69E-03	1.11E-04	1.49E-04	2.37E-05
MEP	kg N eq.	1.03E-01	1.36E-03	2.16E-03	1.41E-03
Ecotox	CTUe	1.76E+01	1.18E+01	2.75E+00	4.90E+00

[a] Elementary flows for this system were derived from Basset-Mens et al. (2009) and feed requirements of animals from DairyNZ (2014b); [b] Elementary flows for these systems are presented in Table D1. Environmental emissions were inventoried using inventory models and key assumptions, as presented in Table D2, and the proxy flows are presented in Table D3.

Table D5 Fossil energy and electricity used in NZ dairy farming systems. Data were averaged across dairy farming in the North Island, New Zealand.

Energy sources	Units	Values
Electricity	MJ/kg milksolids	2.65
Diesel	MJ/kg milksolids	1.71
Petrol	MJ/kg milksolids	0.80

Source: StatisticsNZ (2014).

Table D6 Response of pasture production (on a dry matter [DM] basis) to different application rates of nitrogen (N) fertiliser.

Nitrogen fertiliser rate (kg N per ha per year)	Pasture response (kg DM/kg N)
111-230	16.8
231-363	16.6
364-502	15.2
503-627	12.3

Data based on model of Li et al. (2011).

Table D7 Elementary flows associated with the two beef production systems in New Zealand. Data (annual) were derived from Lieffering et al. (2012) for Farm Class 5 (North Island intensive finishing) where the ratio of beef to sheep was highest compared with other Farm Classes (Beef+LambNZ 2015). Based on these data, two theoretical beef farm systems were generated: (i) in which a mix of beef derived from culled and fattened animals was the only product; and (ii) in which live surplus calves and a mix of beef were the co-products, and in order to account for environmental impacts associated with surplus calves in the latter system, environmental emissions for the co-product beef were handled using system substitution, and assumed to displace a mix of beef in the former farm system.

	Units	Cow-calf-finishing	Cow-calf
Farm inputs			
• Effective farmland [a]	ha	1	1
• Breeding cows	#	0.84	1.41
• 2nd year heifers	#	0.34	0.25
• 2nd year steers	#	0.34	-
• 1st year heifers	#	0.34	0.25
• 1st year steers	#	0.34	-
• Bulls	#	0.001	0.003
• Nitrogen (N) fertilisers	kg N	17	17
• Phosphorus (P) fertilisers	kg P	14	14
• Electricity	kWh	12.61	12.61
• Diesel	L	13.77	13.77
• Petrol	L	10.12	10.12
• Herbicides	kg active ingredients	0.17	0.17
• Grass seeds	kg	0.56	0.56
• Clover seeds	kg	0.07	0.07
Farm outputs			
• Total meat production	kg	148	42

○ Culled cows	kg	24.5	41
○ Culled bulls	kg	0.5	1
• Finishing heifers	kg	46	-
• Finishing steers	kg	77	-
• Surplus calves	#	-	0.89

[a] Adjusted farmland to exclude sheep production system according to Wiedemann et al. (2015).

Table D8 Multiple environmental impacts associated with the two beef production systems and the production of maize silage in New Zealand, calculated on a per-kg of their functional units (FU). The impacts were based on a cradle-to-farm gate perspective for individual systems.

Impacts	Units	Cow-calf-finishing (FU = 1 kg live weight)	Cow-calf (FU = 1 kg live weight)	Maize system (FU = 1 kg DM maize silage)
CC	kg CO_2 eq.	6.97E+00	5.46E+01	1.43E-01
ODP	kg CFC-11 eq.	4.77E-08	4.19E-07	7.89E-09
Non-cancer	CTUh	2.34E-06	1.95E-05	2.08E-07
Cancer	CTUh	5.11E-08	4.26E-07	4.78E-09
PM	kg $PM_{2.5}$ eq.	3.45E-03	2.13E-02	2.43E-04
IR	kg U^{235} eq.	4.89E-02	4.12E-01	2.12E-03
POFP	kg NMVOC eq.	2.76E-02	2.28E-01	1.84E-03
AP	molc H^+ eq.	1.29E-01	7.35E-01	7.18E-03
TEP	molc N eq.	5.66E-01	3.21E+00	3.20E-02
FEP	kg P eq.	1.96E-03	1.63E-02	2.37E-05
MEP	kg N eq.	3.45E-02	2.84E-01	1.41E-03
Ecotox	CTUe	6.10E+00	5.05E+01	4.90E+00

Table D9 Indirect Land use Change accounting methods used in the present study.

Methods	Explanation
Schmidt et al. (2015)	Schmidt et al. (2015) classified sources of land use into three main categories: (i) use of land that is already in use, (ii) intensification of existing land in use, and (iii) transformation of new land. To simplify assessment, the present study focused only on the transformation of new land and assumed that (i) new pasture required for displaced beef pasture was transformed from natural grassland and (ii) displaced cropland was transformed from secondary forest, and their CC emission factors were 590 and 2,070 kg CO_2 eq. per hectare per year, respectively (Schmidt and Muños 2014).
Tonini et al. (2015)	Tonini et al. (2015) estimated emission factors to assess ILUC impacts resulting from the expansion of arable land, at the expense of deforestation for individual regions (biomes) across the world, using a deterministic model. The concept for this approach is associated with the loss of opportunity to accumulate carbon from normal growth of natural forests (vegetation) as a result of deforestation (i.e. the ecosystem cannot perform its normal function). The CC emission factor for increased demand for one hectare of arable land derived from using a weighted-average across the world's regions was 4,100 kg CO_2 eq. per hectare per year. In order to deploy this approach, it was assumed that cumulative extra demand for land (pasture plus cropland) in the present study was arable land.

DRC 16

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STATEMENT OF CONTRIBUTION
TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Jeerasak Chobtang

Name/Title of Principal Supervisor: Sarah McLaren

Name of Published Research Output and full reference:

Chobtang, J., McLaren, S.J., Ledgard, S.F. and Donaghy, D.J. 2016. Consequential Life Cycle Assessment of pasture-based milk production: a case study in the Waikato region, New Zealand. Journal of Industrial Ecology (accepted).

In which Chapter is the Published Work: Chapter 5

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate:

  and / or

- Describe the contribution that the candidate has made to the Published Work:

  The candidate was responsible for developing an intensification framework for increased milk production, performing environmental assessment, interpreting the results and writing the majority of the manuscript.

Jeerasak Chobtang

Candidate's Signature

12 May 2016

Date

Sarah McLaren

Principal Supervisor's signature

16 May 2016

Date

GRS Version 3– 16 September 2011

Chapter 6 Prospective Attributional Life Cycle Assessment of Pasture-Based Milk Production Systems in the Waikato Region, New Zealand

This chapter has been modified and submitted for publication in Journal of Cleaner Production.

Abstract

The predicted increase in global demand for dairy products over the next decade drives the global dairy sector to increase its production capacity. For the New Zealand dairy sector, farm intensification is a common strategy to increase milk production per hectare, and is generally achieved through increased stocking rate, supported by different feed provision methods. In the present study, seven prospective (for the year 2025) scenarios were established, and multiple environmental impacts of these scenarios were assessed using prospective attributional Life Cycle Assessment (LCA). Increased per-animal productivity, and increased pasture utilization efficiency reduced all environmental indicators per kg of standardized milk produced. Increased use of locally produced maize silage reduced Climate Change, Ionizing Radiation, Freshwater Eutrophication Potential and Marine Eutrophication Potential but increased Ecotoxicity for Aquatic Freshwater. The use of imported wheat grain increased all environmental indicators (except Climate Change and Terrestrial Eutrophication Potential), resulting mainly from the production and long-distance transport of wheat grain. In conclusion, scenario-based prospective attributional LCA assists in highlighting the environmental trade-offs and hotspots among prospective farm intensification scenarios in pasture-based dairy systems as well as facilitating decision-making associated with the choice of most environmentally-preferable future dairy systems.

Keywords Animal productivity • Environmental impact • Intensification methods • Life Cycle Assessment (LCA) • New Zealand • Pasture-based dairy system

6.1 Introduction

The predicted global demand for dairy products in 2024 will be approximately 30% more than that in 2014 (OECD/FAO 2015). In order to support such a demand, the global dairy sector will have to increase its production capacity. For New Zealand, which is the single largest dairy exporter in the world (New Zealand Government 2012), an increase in milk production will be associated largely with farm intensification, i.e. increased milk production per hectare (MacLeod and Moller 2006).

The dairy sector in New Zealand is the single largest contributor to income from exports in the nation, accounting for ~25% of the total national export revenue in 2012 (New Zealand Government 2013). The New Zealand Government has a clear vision to double income from agricultural exports by 2025, including dairy products (New Zealand Government 2012). As a result, the New Zealand dairy sector is actively investigating how to increase its production capacity to support this national goal. In principle, two approaches can be used to increase total milk production: (i) farm expansion, and (ii) farm intensification. The former requires additional lands to be converted either from other existing land-based agricultural systems (e.g. sheep and beef farming systems, and forestry) or nature (e.g. forest and native grassland), whereas the latter keeps the area of effective dairy farmland constant; although additional land is required to produce additional inputs to support intensification (e.g. extra cows and brought-in feeds). The present study focused on farm intensification since farm expansion is likely to be limited by restricted availability of new agricultural land in New Zealand.

Generally, intensification methods in pasture-based dairy systems are associated with increased stocking rate coupled with the increased use of off-farm inputs in order to maximize milk production per hectare of dairy farmland (MacLeod and Moller 2006; Pinares-Patiño et al. 2009). The use of various off-farm inputs is largely related to the provision of feed to support an increased number of animals. However, different intensification methods can result in different environmental impacts of milk production (de Léis et al. 2014; O'Brien et al. 2014). Therefore, the potential environmental benefits and disadvantages of these methods need to be comprehensively assessed in order to ensure the overall environmental sustainability of future pasture-based milk production systems. Life Cycle Assessment (LCA) is a holistic analytical approach that can be used to assess multiple environmental impacts of alternative product systems (extending along the life cycle from the acquisition of raw material through to manufacturing, use and on to end-of-life); use of this approach reveals the differences in environmental impacts between these alternative systems, and also any burden-shifting (i.e. the situation where an activity decreases environmental impacts at one (or more) life cycle stage(s) but unintentionally increases other environmental impacts at other life cycle stages) (Finnveden et al. 2009; Hellweg and Milà i Canals 2014).

Based on the literature study (Chapter 2), there are no published studies on multiple environmental impacts of prospective (future) dairy farming systems using an LCA approach. Therefore, the present study aimed to: (i) assess multiple environmental impacts, and (ii) investigate potential environmental trade-offs associated with future pasture-based dairy systems with different intensification methods.

6.2 Methods

The basic concept of attributional LCA (Finnveden et al. 2009) was followed to estimate the environmental impacts associated with different future pasture-based dairy systems. In principle, this approach was compliant with the standardized LCA framework (ISO 14040 and 14044) defined by the International Organization for Standardization (International Organization for Standardization 2006a, 2006b). The dairy-specific methodologies developed for LCA of dairy products by the International Dairy Federation (IDF) (International Dairy Federation 2010) were also used (e.g. co-product handling methods and functional unit). Prospective dairy farming scenarios were assumed to have reached 50% more milk production per hectare than that in 2012 through the use of alternative potential farm intensification methods. The prospective scenarios were based on extrapolation of trends using historical data from the past 10 years (DairyNZ 2014) and the opinions of dairy experts. Environmental emissions of the selected prospective scenarios were modelled using the SimaPro 8 software (Pré Consultants 2013). It should be noted that environmental impacts associated with an average pasture-based milk production system in the Waikato region, New Zealand over one production year (2010/11), previously characterized in Chapter 3, were used as a reference (REF) scenario.

6.2.1 Key assumptions

Two key assumptions were made in the present study: (i) all unit processes involved in prospective dairy farming scenarios were already in place, and (ii) changes (either increases or decreases) in demand for all supporting products and their intermediate flows were not allowed. Other assumptions associated with life cycle inventory procedures and climatic conditions (e.g. pattern of rain events and total precipitation) for the prospective scenarios were the same as the 30-year average in the Waikato region.

6.2.2 Selection of prospective scenarios

Use of scenario analysis in LCA studies has been extensively described (Höjer et al. 2008; Pesonen et al. 2000; Weidema 2003). In the present study, a normative (goal-oriented) scenario was adopted, referring to a well-defined target to be reached by adjusting or adapting an existing product system (Höjer et al. 2008; Weidema 2003). The prospective milk production target was assumed to be reached by using different farm intensification methods (e.g. the use of different off-farm inputs such as brought-in feeds and nitrogen [N] fertilisers) or farm management practices (e.g. increased pasture utilization efficiency).

As a first step in the analysis, key aspects of dairy farming (e.g. milk production per hectare and milk production per cow) were extrapolated using historical data over the past 10 years (1995 – 2014) (DairyNZ 2014). This was regarded as a 'business as usual' (BAU) trend, and assumed an ongoing linear increase in milk production. Extrapolation of this trend indicated that the BAU increase would lead to a 30% increase in milk production in 2025 relative to 2012, implying that an extra 20% milk production would have to be produced in order to reach the desired target of a 50% increase. This gap was thus assumed to be filled by extra milk production which was derived from different farm intensification methods, and farm intensification options were focused on either: (i) increased animal productivity (more milk per cow), or (ii) increased stocking rates (more cows per hectare) (MacLeod and Moller 2006; Pinares-Patiño et al. 2009); these two intensification options generally result in increased milk production per hectare. The extra milk production for each scenario was assumed to be supported by provision of extra feed supply which was derived either from: (i) on-farm production (e.g. improved pasture use efficiency or increased use of N fertilisers to boost extra pasture production), or (ii) brought-in feed sources. Note that these farm practices are increasingly common in pasture-based dairy systems in New Zealand (Clark et al. 2013; Ramsbottom et al. 2015).

Seven potential prospective dairy farming scenarios in the Waikato region, New Zealand were defined and are summarized in Table 6.1. A detailed description of these scenarios is given in Section E1 of the Supplementary material E.

Table 6.1 Characteristics of the selected scenarios for future pasture-based dairy farming systems in the Waikato region, New Zealand.

Scenarios [a]	BAU trend [b]	Meeting additional 20% of extra milk production through [d]:					
		(i) Improved animal productivity [c]	(ii) Increased stocking rate coupled with different feed provision methods				
			Increased use of mixed brought-in feed	Increased pasture utilization efficiency	Increased use of nitrogen fertiliser to boost on-farm pasture production	Increased use of brought-in maize silage	Increased use of imported wheat grain
REF							
BAU	√						
ALT 1	√	√					
ALT 2	√		√				
ALT 3	√		√	√			
ALT 4	√		√		√		
ALT 5	√		√			√	
ALT 6	√						√

[a] A detailed description of each scenario is given in Section 6.5.1; [b] all prospective scenarios were assumed to produce 30% more milk per hectare relative to the reference scenario (REF scenario); [c] additional demand for feed in this scenario was supported by increased use of mix brought-in feed; [d] additional 20% of milk production for the ALT 1-6 scenarios was assumed to be derived from the use of different farm intensification methods in order to meet a desired milk production target of 50% in 2025 relative to that in 2012.

Estimated farm inputs and outputs for the selected prospective farm scenarios are presented in Table 6.2. As the scenarios to increase milk production to reach the 2025 target were achieved through farm intensification, the effective dairy farmland area was kept unchanged. In the ALT 1 scenario, stocking rate was set lower than that in other scenarios, because individual cows would produce more milk than in the other scenarios. For the ALT 2-6 scenarios, extra milk production was achieved through an increased stocking rate, which was supported by increasing the amount of feed. For the ALT 2-5 scenarios, the amount of mixed brought-in feeds was increased, at a constant ratio of 60% palm kernel expeller, 33% maize silage, 6% conserved pasture and 1% concentrate feed, on a dry matter basis (see Chapter 3 for more details). In the ALT 3-5 scenarios, extra feed intake was provisioned through: (i) increased pasture utilization efficiency (i.e. animals ingested more feed from existing pastures), (ii) increased use of N fertiliser (urea) to boost extra pasture production which was assumed to be consumed by animals, and (iii) increased use of brought-in maize silage, respectively. In the ALT 3 and 4 scenarios, increased pasture intake resulted in a reduction in total amount of brought-in feed compared with other prospective scenarios involving increased stocking rate. For the ALT 6 scenario, all brought-in feed was replaced by wheat grain imported from a global market. It should be noted that the actual amounts of feed offered exceeded that for the amount of total feed ingested due to accounting for feed lost during feeding-out (i.e. offering feed to animals), as recommended by DairyNZ (2015).

Table 6.2 Estimated farm inputs and outputs of the prospective dairy farming scenarios. In all cases, data are presented on a per hectare basis over one production year, and a per-ha basis refers to the on-farm stage only (i.e. area used for grazing and milking dairy cows).

Farm traits	Units	Scenarios [j]							
		REF	BAU	ALT 1	ALT 2	ALT 3	ALT 4	ALT 5	ALT 6
General farm traits									
• Total farmland [a]	ha/herd	165	165	165	165	165	165	165	165
• Total milking cows	cow/herd	448	553	553	639	639	639	639	639
• Replacement rate [b]	%	23	23	23	23	23	23	23	23
• Culling rate [b]	%	23	23	23	23	23	23	23	23
• Calving rate [b]	%	80	80	80	80	80	80	80	80
• Stocking rate [c]	cow/ha	2.73	3.35	3.35	3.87	3.87	3.87	3.87	3.87
• Animal productivity [c]	kg FPCM/cow	4,730	5,508	6,357	5,508	5,508	5,508	5,508	5,508
Farm inputs									
• Chemical fertilisers [d]									
○ Nitrogen (N)	kg N/ha	142	142	142	142	142	279	142	142
○ Phosphorus (P)	kg P/ha	49	49	49	49	49	49	49	49
○ Potassium (K)	kg K/ha	45	45	45	45	45	45	45	45
• Fossil energy [d]									
○ Diesel	MJ/kg FPCM	0.158	0.158	0.158	0.158	0.158	0.158	0.158	0.158
○ Petrol	MJ/kg FPCM	0.074	0.074	0.074	0.074	0.074	0.074	0.074	0.074
○ Lubricant oil	MJ/kg FPCM	0.002	0.002	0.002	0.002	0.002	0.002	0.002	0.002
• Electricity [d]	MJ/kg FPCM	0.244	0.244	0.244	0.244	0.244	0.244	0.244	0.244
Total feed intake [e]	kg DM/ha	11,744	16,333	17,489	18,608	18,608	18,608	18,608	17,101
• Estimated pasture intake [f]	kg DM/ha	9,000	9,000	9,000	9,000	11,275	11,275	9,000	9,000
• Intake of brought-in feeds [g]	kg DM/ha	2,744	7,333	8,489	9,608	7,333	7,333	9,608	8,101
○ Palm kernel expeller	kg DM/ha	1,643	4,400	5,093	5,765	4,400	4,400	4,400	-
○ Maize silage	kg DM/ha	893	2,420	2,801	3,171	2,420	2,420	4,695	-
○ Conserved pasture	kg DM/ha	276	440	509	576	440	440	440	-
○ Concentrate	kg DM/ha	34	73	85	96	73	73	73	-
○ Wheat grain	kg DM/ha	-	-	-	-	-	-	-	8,101
Farm outputs [h]									
• Milk production [i]	kg FPCM/ha	12,851	18,456	21,296	21,296	21,296	21,296	21,296	21,296
• Culled cows	cow/ha	0.63	0.77	0.73	0.89	0.89	0.89	0.89	0.89
• Surplus calves	#/ha	1.55	2.37	2.24	2.74	2.74	2.74	2.74	2.74

[a] Increased milk production was achieved through farm intensification (keeping dairy farmland constant) which was achieved by either: (i) increased milk production per cow (animal productivity), or (ii) increased stocking rate; [b] these farm traits were assumed to be constant and derived from average dairy farms in the Waikato region for the production year 2010/11 (see Chapter 3 for more details); [c] these parameters were extrapolated from data over the past 10 years (2005-2014); [d] for all scenarios, data were derived from Chapter 3, except for the ALT 4 scenario where extra milk production was associated with increased use of N fertilisers to boost extra pasture production; [e] total feed intake = estimated pasture intake + total intake of brought-in feeds; [f] pasture intake was calculated using inventory methods according to the New Zealand Ministry for Primary Industries (2013); [g] estimation of intake of brought-in feeds was derived from Chapter 3, and the proportion of feed types was assumed to be unchanged; [h] estimation of milk production was extrapolated using data from 2005 – 2014, derived from DairyNZ (2014); estimation of cull cows and surplus calves was computed using calving rate, culling rate and replacement rate (see Chapter 3 for more details); [i] milk production was estimated to be 50% greater per hectare than 2012 data for the Waikato region of 14,197 kg of fat- and protein-corrected milk (FPCM) per ha (DairyNZ 2014); [j] detailed description of the different farming scenarios is given in Section E1 of the Supplemtary material E.

6.2.3 Scope definition

Environmental emissions associated with farm infrastructure (e.g. farm buildings, machinery, fences and roads), animal medicines and reproduction-related activities in the foreground processes (i.e. both off-farm and on-farm stages) were not accounted for, due to lack of associated data.

6.2.3.1 System boundary

A cradle-to-farm gate perspective was used as the system boundary. The simplified flows starting from acquisition of raw material through to milk production at the farm gate are depicted in Figure 6.1. Detailed information associated with energy and material flows is described in Section E2 of the Supplementary material E.

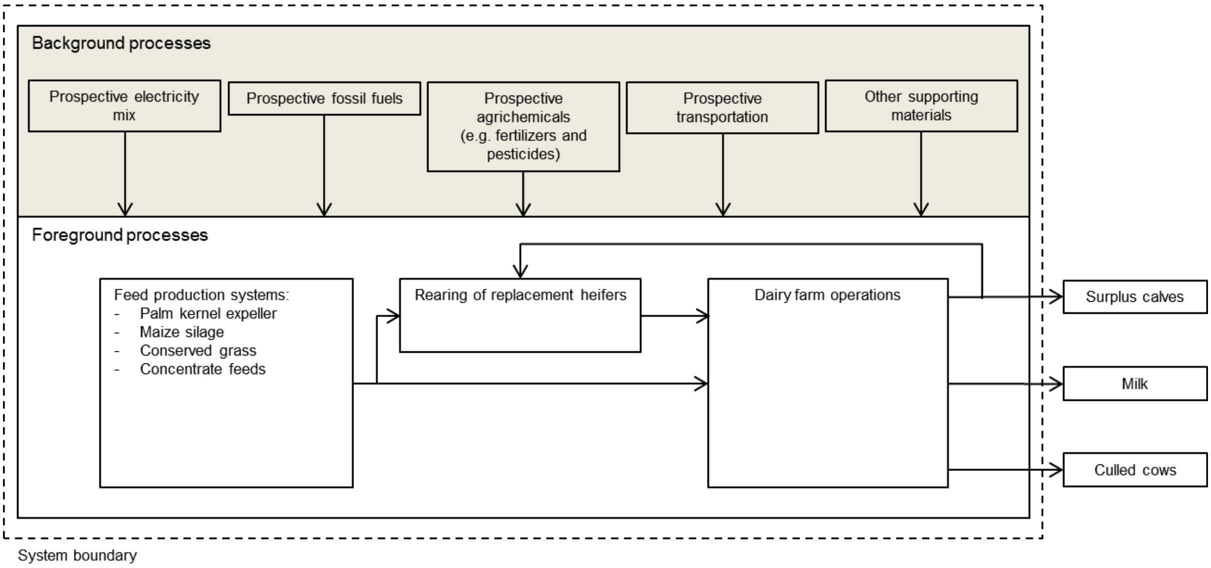

Figure 6.1 Simplified flows and system boundary for the cradle-to-farm gate life cycle of prospective pasture-based dairy farming systems.

6.2.3.2 Functional unit

One kg of fat- and protein-corrected milk (FPCM) was used as a functional unit, as recommended by the International Dairy Federation (IDF) (International Dairy Federation 2010): FPCM (kg) = milk yield (kg) × [(0.1226 × fat%) + (0.0776 × protein%) + 0.2534].

6.2.3.3 Allocation methods

In the cradle-to-farm gate life cycle of pasture-based dairy production systems, there are some unit processes that produce multiple products (e.g. grain and straw for feed crop systems, milk and meat for dairy systems). In the present study, the co-products were handled according to the IDF guidelines (International Dairy Federation 2010). The IDF guidelines recommended allocating environmental burdens between co-products associated with farm inputs (e.g. feed) based on their 5-year average economic values, whereas the allocation method recommended for co-products associated with farm outputs (i.e. milk and meat) was based on its biophysical relationship (International Dairy Federation 2010).

6.2.4 Life cycle inventory analysis

The inventory methods used in the present study were similar to those in Chapter 3. The exception was that all energy and material flows were replaced by prospective technology mixes. The prospective technology mixes included prospective electricity mix (see Section E3 of the Supplementary material E), and prospective fertilisers, lime and pesticides (see Section E4 of the Supplementary material E). For prospective fossil energy (e.g. diesel, petrol and lubricant oils) as well as road (Euro 5 engine specification) and transoceanic transportation, data were derived from the ecoinvent v3 database (ecoinvent Centre 2013).

In the ALT 4 scenario where increased feed intake was derived from increased pasture production resulting from increased use of N fertiliser, estimated pasture production in response to N fertiliser applied (kg dry matter (DM) per kg N) was modelled according to Li et al. (2011), as shown in Section E5 of the Supplementary material E. It should be noted that average pasture N concentration in New Zealand dairy systems was used throughout the study (3.7% N, on a DM basis; see Kolver (2000)). In addition, for the ALT 4 scenario, extra prospective lime was used to neutralize potential acidification of pastoral soils, as recommended by Sinclair et al. (1993), of 0.5 kg of lime per kg of N.

6.2.5 Impact assessment methods

The study included multiple environmental impacts in order to identify possible burden-shifting between impact categories. Twelve out of 15 impacts categories which were characterized by the most up-to-date characterization models (Hauschild et al. 2013), as recommended by the European Food SCP Roundtable (2013) were selected; these are presented in Table 6.3.

Table 6.3 Impact categories considered in the present study, based on the European Food SCP Roundtable (2013). The impacts superscripted by * were not included in the present study due to a lack of relevant inventory data.

Impact category	Units [a]	Abbreviation
Climate Change	kg CO_2 eq.	CC
Ozone Depletion Potential	kg CFC-11 eq.	ODP
Human Health Toxicity, non-cancer effects	CTU_h	Non-cancer
Human Health Toxicity, cancer effects	CTU_h	Cancer
Particulate Matter	kg $PM_{2.5}$ eq.	PM
Ionizing Radiation – human health effects	kBq U^{235} eq.	IR
Photochemical Ozone Formation Potential	kg NMVOC eq.	POFP
Acidification Potential	molc H^+ eq.	AP
Eutrophication – terrestrial ecosystems	molc N eq.	TEP
Eutrophication – freshwater ecosystems	kg P eq.	FEP
Eutrophication – marine ecosystems	kg N eq.	MEP
Ecotoxicity for Aquatic Freshwater	CTU_e	Ecotox
Resource Depletion-mineral, fossil*	kg antimony (Sb) eq.	RD (mineral, fossil)
Resource Depletion-water use*	m^3 water use related to scarcity of water	RD (water use)
Land Use Impact*	kg carbon (C) deficit	LU

[a] CO_2 = carbon dioxide; eq. = equivalent; CFC-11 = trichlorofluoro-methane; CTU_h = comparative toxic unit for humans; $PM_{2.5}$ = particulate matter less than 2.5 μ in diameter; kBq = kilobecquerel; U^{235} = uranium-235; NMVOC = non-methane volatile organic compounds; molc = mole of charge; H^+ = hydrogen ion; N = nitrogen; P = phosphorus; CTU_e = comparative toxic unit for ecosystems.

6.2.6 Contribution analysis

A contribution analysis assists in the interpretation of LCA results with regard to identifying environmental hotspots along the life cycle of a product (Heijungs et al. 2005). In the present study, relevant flows in the cradle-to-farm gate life cycle of prospective pasture-based milk production systems were aggregated to different life cycle stages, as follows:

- The on-farm stage (Onfarm): This stage included emissions associated with on-farm activities (e.g. milking cows), the use of chemical fertilisers and pesticides, the use of fossil fuels and electricity, as well as management of farm effluent and animal excreta. Emissions associated with production of prospective fossil fuels and electricity were included in this stage.

- The rearing of replacement animals (Replacement): This stage covered emissions associated with the production of prospective replacement heifers, i.e. from about 4 days of age up to nearly calving. For all prospective scenarios, this stage was assumed to be carried out off-farm. It should be noted that environmental emissions associated with the manufacturing of agrichemicals, brought-in feed and transportation of off-farm inputs for use in this sub-system were included in this stage.

- The production of brought-in feeds (Feeds): This stage aggregated emissions associated with the production of prospective brought-in feeds to be used on dairy farms, including palm kernel expeller, maize silage, pasture silage, pasture hay and concentrate. A mix of 0.6 maize grain and 0.4 barley grain was used as a proxy for concentrate feed. Environmental emissions associated with the manufacturing of agrichemicals and transportation of off-farm inputs for use in each of these sub-systems were included in this life cycle stage.

- The manufacturing of agrichemicals (Agrichemicals): This stage included emissions originating from the prospective manufacturing of chemical fertilisers, lime and pesticides to be used on dairy farms.

- The transportation (Transport): Environmental emissions associated with transportation of prospective off-farm inputs, including brought-in feeds, agrichemicals and fossil fuels, to dairy farms were summed in this stage.

6.2.7 Sensitivity analysis

A sensitivity analysis helps in determining influential factors associated with choices (e.g. parameter inputs, system boundaries and allocation methods) and assumptions that have significant influences on final LCA results (Heijungs and Guinée 2007; Huang et al. 2013).

In the present study, sensitivity analyses were carried out to investigate effects of the choices associated with: (i) the manufacturing of urea, (ii) the effect of maize silage genetic development and (iii) the use of system substitution to handle dairy co-products.

Manufacturing of liquid ammonia (i.e. as a precursor for urea and other N fertilisers) is a resource, energy and emission intensive process (Hasler et al. 2015). Therefore, an improvement in manufacturing of fertilisers (e.g. improved energy use efficiency and reduced environmental emissions) is expected to help reduce environmental impacts (Rafiqul et al. 2005). In principle, there are two major manufacturing pathways to produce liquid ammonia: (i) partial oxidation, and (ii) steam reforming (Rafiqul et al. 2005). While heavy fuel oil is used as the main precursor for hydrogen production in the partial oxidation pathway, natural gas is predominately used in the steam reforming pathway (Rafiqul et al. 2005).

Three options associated with manufacturing technologies of urea fertiliser were tested (Table 6.4). The first option (Option I) was urea manufactured using the partial oxidation pathway in China where the prospective Chinese electricity grid mix (i.e. large share of coal-fired power plants; ~65% of the total) was linked. The second option (Option II) was urea manufactured using the steam reforming pathway in The Netherlands where the prospective Dutch electricity was linked. The third option (Option III) was urea manufactured as in Option II but with a 25% improvement in energy use efficiency (i.e. the demand for natural gas and electricity and relevant emissions were assumed to be reduced by 25%). In addition, in Option III, urea was assumed to be manufactured in New Zealand using electricity solely generated from wind power. Multiple environmental impacts of these N fertilisers are presented in Section E6 of the Supplementary material E. In the sensitivity analysis, these flows were tested by replacing the urea flow in all sub-systems in the foreground processes in the ALT 4 scenario.

Table 6.4 Sensitivity analysis associated with choices of nitrogen (urea) fertilisers.

Country of manufacture	Pathway [a]	Main precursor [c]	Electricity [d]	Distance (km)	
				International	National
China (CN)	PO	Heavy fuel oil	CN grid mix	11,000	300
The Netherlands (NL)	SR	Natural gas	NL grid mix	21,000	300
New Zealand (NZ)	Improved SR [b]	Natural gas	NZ wind power	-	300

[a] PO = partial oxidation pathway and SR = steam reforming pathway; [b] assuming that energy efficiency in this process was increased by 25% (all inputs, except manufacturing plants that were kept constant), and all environmental emissions were reduced by 25%; [c] the main precursor for producing hydrogen for use as a substrate in manufacturing process of liquid ammonia; [d] Chinese electricity grid mix was derived mainly from coal-fired power plants (accounting for ~65% of the total), Dutch electricity generation was based largely on natural gas (accounting for ~49% of the total), and a wind power plant in New Zealand was assumed to be 1-3 MW capacity, and to be the only source (100%) of electricity generation to support a New Zealand urea manufacturing plant.

A sensitivity analysis was also carried out to test the relative effects of improvement in maize genetics where an increase in silage maize production was expected, based on data from trends over 10 years by a major maize company in New Zealand (R. Densley, personal communication). Based on extrapolation from this past research, over the next 10 years, silage maize is likely to produce an extra 3,100 kg DM per hectare (or 1.3% improvement in total DM) compared with the current yield (23,000 kg DM per hectare). The analysis was carried out only on the ALT 5 scenario where extra milk production was derived from the use of extra maize silage.

The use of system substitution to handle dairy co-products (dairy meat and live surplus dairy calves) was tested. These two dairy co-products were assumed to displace (substitute for) beef and live surplus beef calves derived from conventional beef systems. Generally, the market of conventional beef systems has been considered as global since beef can be traded internationally (Dalgaard et al. 2014). However, comprehensive environmental data for a global beef mix (e.g. weighted-average across major beef exporting nations) does not exist. Therefore, in the present study, the average conventional beef system in New Zealand was used as a proxy. The New Zealand conventional beef farm model (see Chapter 5, section 5.2.2.7) was inventoried and modified by replacing the flows with the corresponding prospective flows, e.g. future electricity grid mix, fertilisers, agrichemicals and transport (see Sections E4 and E5). Six prospective scenarios (Scenarios ALT 1 – 6) were evaluated. It should be noted that the replacement (substitution) rates for both dairy co-products were assumed to be 1:1 on a mass basis.

6.3 Results and discussion

The results revealed environmental trade-offs across the different prospective dairy farming scenarios. The trade-offs were associated mainly with shifts in environmental impacts between the life cycle stages and in the relative magnitude of results for different impact categories.

6.3.1 Environmental profiles

The environmental profiles for the different prospective scenarios are depicted in Figure 6.2. The absolute values for individual impact categories for each prospective scenario and the contribution of its life cycle stages are presented in Section E7 of the Supplementary material E.

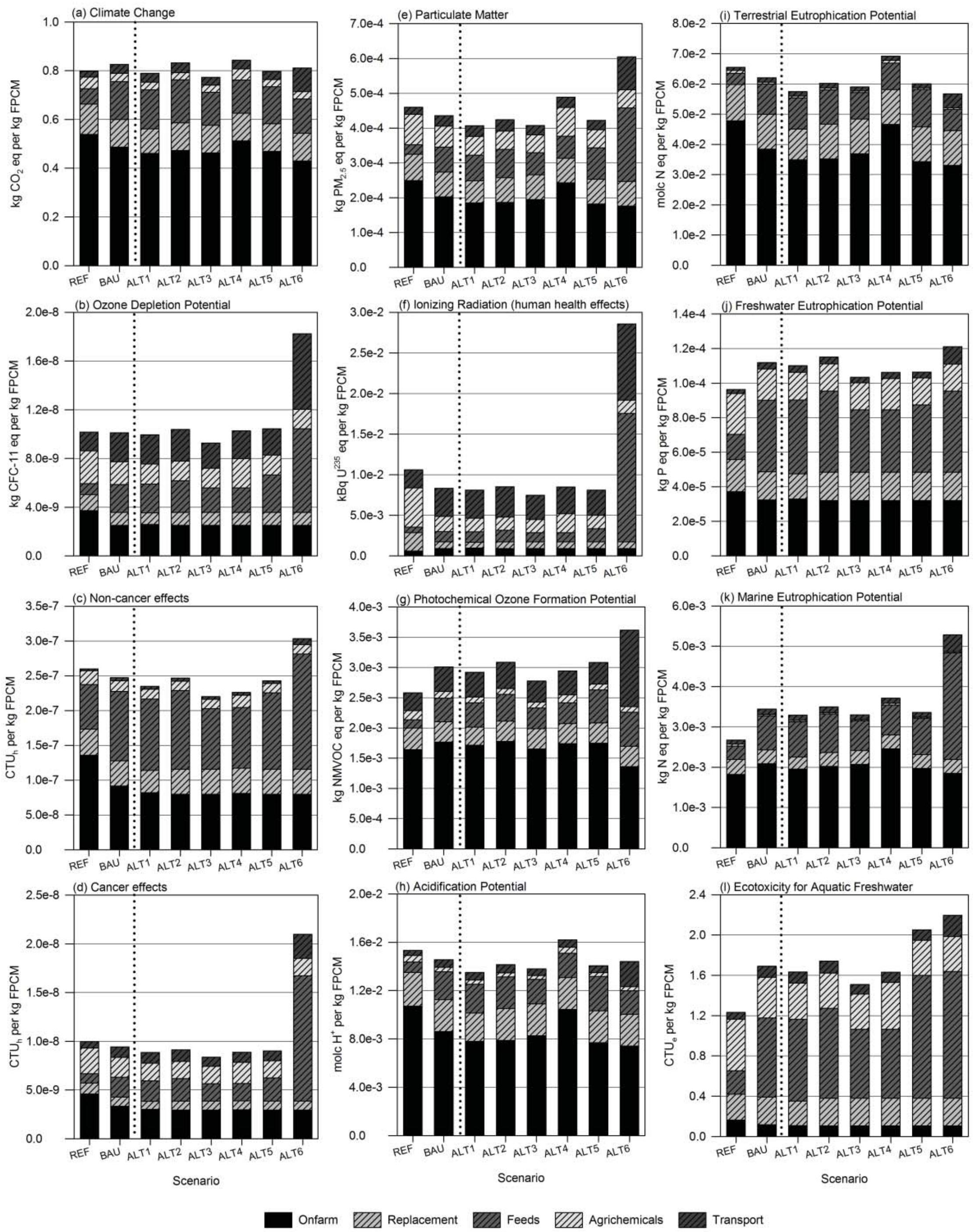

Figure 6.2 Environmental impacts [(a) Climate Change, (b) Ozone Depletion Potential, (c) Human Health Toxicity (non-cancer effects), (d) Human Health Toxicity (cancer effects), (e) Particulate Matter, (f) Ionizing Radiation (human health effects), (g) Photochemical Ozone Formation Potential, (h) Acidification Potential, (i) Terrestrial Eutrophication

Potential, (j) Freshwater Eutrophication Potential, (k) Marine Eutrophication Potential and (l) Ecotoxicity for Aquatic Freshwater] associated with reference scenario (REF) and prospective scenarios [BAU = extrapolated dairy farming system in the Waikato region in 2025, ALT 1 = improved milk production per cow coupled with the use of brought-in feeds, ALT 2 = increased stocking rate coupled with the use of brought-in feeds, ALT 3 = increased stocking rate coupled with improved pasture utilization efficiency, ALT 4 = increased stocking rate coupled with extra pasture production through the use of N fertiliser, ALT 5 = increased stocking rate coupled with use of extra brought-in maize silage and ALT 6 = increased stocking rate coupled with the use of brought-in wheat grain], and the contribution of different life cycle stages [Onfarm = the on-farm stage, Replacement = the rearing of replacement heifers, Feeds = the production of brought-in feeds, Agrichemicals = The manufacturing of agrichemicals for use on dairy farms, and Transport = the transportation of off-farm inputs for use on dairy farms].

For the purpose of comparative analysis, the results from two other recently-published scientific papers (Battini et al. 2014; Battini et al. 2015) that used the same LCA approach (i.e. attributional LCA) and the same impact assessment methods to assess the environmental impacts of confinement-based dairy farming systems in Italy were compared with the range of impacts for the prospective dairy scenarios in the present study (presented in [brackets] in the following discussion). First, environmental impacts associated with Italian dairy farming systems where four levels of farm intensification were compared, considering one farm to represent each intensification level, and using attributional LCA with biophysical allocation for milk and meat (Battini et al. 2015). The multiple environmental indicators calculated in their study, on a per-kg FPCM, across the four farm systems ranged from 1.02 – 1.26 [0.77 – 0.84] kg CO_2 eq. for the impact on CC; 1.00E-02 – 1.11E-02 [1.35E-02 – 1.62E-02] molc H^+ eq. for the impact on AP; 1.20E-04 – 1.90E-04 [1.03E-04 – 1.21E-04] kg P eq. for the impact on FEP; and 0.79E-02 – 1.33E-02 [3.29E-03 – 5.28E-03] kg N eq. for the impact on MEP. Second, environmental impacts for one large-scale confinement-based dairy farm in Italy were assessed, using attributional LCA with an economic allocation for milk and meat (Battini et al. 2014). Their findings showed that the environmental impacts, on a per-kg FPCM, of this farm system were 1.13 kg CO_2 eq. for the impact on CC; 1.23E-02 molc H^+ eq. for the impact on AP; 1.15E-04 kg P eq. for the impact on FEP; 0.88E-02 kg N eq. for the impact on MEP; 2.64E-03 [2.77E-04 – 3.62E-04] kg NMVOC eq. for the impact on POFP; and 3.88E-04 [4.07E-04 – 6.04E-04] kg $PM_{2.5}$ eq. for the impact on PM.

It should be noted that direct comparison and attributing differences to farm system type is unwise since they are single examples of each type of farm system, and farm practices can vary greatly between and within systems. Additionally, the differences in inventory procedures and assumptions made can influence the final LCA results of life cycle of dairy farming systems.

6.3.2 Environmental trade-offs

The environmental trade-offs are presented as relative changes (%) in environmental indicators between a selected pair of dairy farming scenarios (Table 6.5). The relative comparison of multiple environmental impacts was focused on: (i) the BAU scenario versus the reference (REF) scenario, in order to investigate changes in environmental impacts between future and base dairy systems, (ii) increased animal productivity (ALT 1 scenario) versus increased stocking rate (ALT 2 scenario), to investigate environmental benefits of improved animal productivity, and (iii) scenarios associated with different feed provision methods (ALT 3 – 6 scenarios) versus ALT 2, in order to investigate the contribution of the potential feed provision methods compared with the use of mixed brought-in feeds to support increased stocking rate in pasture-based dairy systems.

Table 6.5 The relative changes (%) in environmental performance between a pair of dairy farming scenarios.

Scenarios [a, b] (*x*/*y*)	CC [c, d]	ODP	Non-cancer	Cancer	PM	IR	POFP	AP	TEP	FEP	MEP	Ecotox
BAU/REF	3	-1	-5	-6	-5	-21	17	-5	-5	17	29	37
ALT 1/ALT 2	-5	-4	-5	-3	-4	-5	-5	-5	-5	-4	-6	-6
ALT 3/ALT 2	-7	-11	-11	-8	-4	-12	-10	-2	-2	-10	-6	-13
ALT 4/ALT 2	1	-1	-8	-3	15	0	-5	15	15	-8	6	-6
ALT 5/ALT 2	-4	0	-2	-1	0	-5	0	-1	0	-8	-4	18
ALT 6/ALT 2	-3	76	23	130	43	235	17	2	-6	5	51	26

[a] See Table E1 for description of these dairy farming scenarios; [b] relative change (%) = $[(x - y)/y \times 100]$; [c] abbreviations for impact categories are described in Table 6.4; [d] the CC indicator result for the reference scenario (see Chapter 3) was re-characterized using the IPCC 2007 characterization factors (Intergovernmental Panel on Climate Change 2007) in order to facilitate a comparison.

6.3.2.1 Comparison between the reference and business-as-usual scenarios

This comparison aimed to investigate the influence of potential future technologies on the net environmental performance of the life cycle of the dairy farming system as opposed to the current technologies (see Section E8 of the Supplementary material E). The results showed that seven environmental indicators in the BAU scenario were relatively lower than those in the REF scenario by 1% to 21%, depending on impact categories) (see Table 6.5), resulting from the improvement in environmental performance of the prospective electricity grid mix as well as ongoing progressive improvement in animal productivity (extrapolated animal productivity in the BAU scenario was ~15% greater than in the REF scenario).

The environmental performance of the prospective electricity grid mix in most countries involved was substantially improved compared with the existing electricity mix (data not shown). This was as a result of an increase in the share of the more environmentally-friendly electricity generation technologies such as hydro, geothermal and wind power plants and the decrease in relatively high emission technologies such as coal-fired power plants (see Section E3). In contrast, five impact indicators (CC, POFP, FEP, MEP and Ecotox) were higher for the BAU scenario compared with the REF scenario. The increase in magnitude of these five indicators was associated with increased use of brought-in feed in the BAU scenario. This implies that environmental improvement in prospective technologies (e.g. electricity) and under-threshold animal productivity (e.g. ~15% in the present case study) cannot guarantee a comprehensive increase in environmentally-sustainable development.

6.3.2.2 Comparison between increased animal productivity and increased stocking rate

This comparison investigated the influence of improved animal productivity on net environmental impacts when compared with increased stocking rate without any improvement in animal productivity (see Table 6.2). The total feed intake (on a DM basis) in the ALT 1 scenario was lower than that in the ALT 2 scenario by 6% due to relatively lower animal maintenance requirement resulting from a lower stocking rate. All environmental indicators assessed in the ALT 1 scenario were relatively lower than those in the ALT 2 scenario (ranging from 3% – 6%, depending on impact categories). The finding confirmed that increased animal productivity can help improve environmental performance of dairy farming systems.

Generally, increased animal productivity (i.e. animals producing more desired outputs, while using the same or less amount of inputs) results in reduced environmental impacts, on a per-unit of output basis, in livestock production systems, including dairy systems (Capper and Bauman 2013; Knapp et al. 2014). This is due partly to the fact that there is a dilution effect where feed requirement for maintenance of milking cows is reduced, while the same amount or more milk is produced (Dijkstra et al. 2013).

6.3.2.3 Comparison of different feed provision methods

The main objective of this comparison was to investigate the effect of different feed provisioning methods on the environmental impact of future dairy systems. The results showed that there were environmental trade-offs between the prospective scenarios where meeting a milk production target was achieved through increased stocking rate coupled with different feeding methods.

Improved pasture utilization efficiency (ALT 3 scenario), which referred to animals harvesting more feed from existing pasture (e.g. McCarthy et al. 2014), showed a relative reduction in total brought-in feeds (see Table 6.2) when compared with the ALT 2 scenario. This led to improved environmental performance through the life cycle of the pasture-based milk production system, ranging from 2% to 13%, depending on impact categories (see Table 6.5). These reductions were associated mainly with the lowered contributions of the production and transportation of brought-in feeds in the ALT 3 scenario. O'Brien et al. (2015) reported that an increased share of grazed grass in total feed intake in pasture-based dairy systems resulted in a reduced carbon footprint of Irish milk.

The provision of increased on-farm feed supply through increased use of N fertiliser to boost extra pasture production (ALT 4 scenario) resulted in environmental trade-offs (see Table 6.5). While the relative reduction in the use of brought-in feeds (as a result of increased pasture consumption) resulted in decreases in the environmental indicators associated with the production (Non-cancer, Cancer, FEP and Ecotox) and transportation (e.g. ODP and POFP) of brought-in feeds, the emissions associated with manufacturing of urea fertiliser (PM) and its use at the on-farm stage (CC, PM, AP, TEP and MEP) led to increases in associated environmental indicators. The increases in environmental indicators at the on-farm stage were driven partly by emissions of nitrogenous substances (e.g. NH_3 and N_2O), resulting from the use of N fertiliser (Ledgard et al. 2009).

The intensification method with increased use of maize silage (ALT 5 scenario) reduced almost all of the environmental indicators assessed in the present study by 1% – 8%, depending on impact category, except for the impact on Ecotox (see Table 6.5). The benefit of using maize silage in a pasture-based dairy system was associated with the dilution effect on N intake since the N concentration in maize silage is relatively low (~1.2% N on a DM basis), leading to reduced urinary N excretion (de Klein et al. 2010; Luo et al. 2010). However, the impact on Ecotox increased (18%), as maize production required phosphate fertiliser (which contained heavy metals) and pesticides (especially insecticides that had a high impact on ecotoxicity for aquatic freshwater) (Berthoud et al. 2011; Yang 2013).

In the ALT 6 scenario where total brought-in feeds were replaced by wheat grain that was assumed to be imported from a global market, almost all environmental indicators increased by 2% – 235%, depending on impact category (see Table 6.5). The increases were associated with increased contributions from the production and long-distance transportation of wheat grain. This was partly because intensive wheat grain production required relatively high inputs, especially phosphate fertilisers (containing heavy metals) and pesticides (e.g. insecticides and fungicides) (ecoinvent Centre 2013), resulting in higher environmental impacts when compared with the production systems of palm kernel expeller and maize silage when computed on a unit of metabolisable energy basis. The exception was for the impacts on CC and TEP, which showed a small decrease (3% – 6%). This is due to wheat grain having a high energy but low protein concentration can reduce enteric CH_4 emissions and lower N ingested, subsequently reducing urinary N excretion and hence NH_3 emissions (Boadi et al. 2004; Gerber et al. 2013). However, its production system as well as its long-distance transportation, which demanded large amounts of resources and released a range of environmental emissions, outweighed its environmental benefits and subsequently resulted in exacerbated environmental impacts of milk.

In confinement-based dairy systems, Battini et al. (2015) reported that there were substantial trade-offs of environmental performance across different levels of farm intensification. For example, higher levels of farm intensification resulted in decreased global environmental impacts per kg milk (e.g. CC), but led to increased local environmental impacts (e.g. FEP) when compared with the lower levels of farm intensification.

6.3.3 Results of sensitivity analyses

Assumptions about the sources of N fertiliser for use in the foreground processes were tested using the ALT 4 scenario. The results showed that a more environmentally-friendly manufacturing pathway for N fertiliser (Option II) resulted in significant reductions in all environmental indicator results (on a per-kg N basis), ranging from 1% to 61%, depending on impact categories, when compared with those from Option I (see Section 10 of the ESM). In addition, improving the efficiency of local N manufacturing linked with electricity derived from wind power (Option III) further lowered environmental impacts ranging from 0% – 60%, depending on impact categories, when compared with Option II (see Section E9 of the Supplementary material E), on a per kg N basis

However, these improvements did not result in substantial reductions in environmental indicator results for the cradle-to-farm gate life cycle of milk, when expressed on a per kg FPCM basis. The reduction in almost all environmental impacts ranged from 0% to 5% for Option II compared with Option I, except for the impacts on ODP (8%), PM (15%) and IR (15%) (see Section E9). Reductions in almost all environmental indicators in Option III ranged from 0% to 4%, depending on impact categories when compared with those in Option II, except for the impacts on ODP (19%) and IR (17%) (see Section E9). In conclusion, these findings highlighted that improved manufacturing efficiency of N fertiliser resulted in substantial reductions in all environmental impacts of urea fertiliser (on a per kg N basis) but did not result in substantially improved environmental impacts in the life cycle of pasture-based milk, on a per kg FPCM basis, except for the impacts where the results were most influenced by manufacturing and transportation of urea fertiliser (e.g. ODP and IR).

For the sensitivity analyses associated with improvement in maize genetics, the results showed that genetic gain in silage maize production had insignificant benefits for all impact categories (less than 1%) for the cradle-to-farm gate life cycle of pasture-based milk production in the Waikato region, except for the impacts on Non-cancer (2%) and Ecotox (5%) when compared with the existing silage maize production system (data not shown).

The comparative results derived from the use of allocation method and system substitution to handle dairy co-products are illustrated in Figure 6.3. As seen in Figure 6.3, most impact indicators assessed for all prospective dairy scenarios derived from the use of system substitution were relatively decreased (except the ODP, IR, Cancer, IR and Ecotox indicators which the magnitudes were relatively unchanged) when compared with the use of allocation method. However, these two co-product handling methods did not alter the ranking of the prospective scenarios. This is mainly because these prospective scenarios produced the same amounts of dairy co-products.

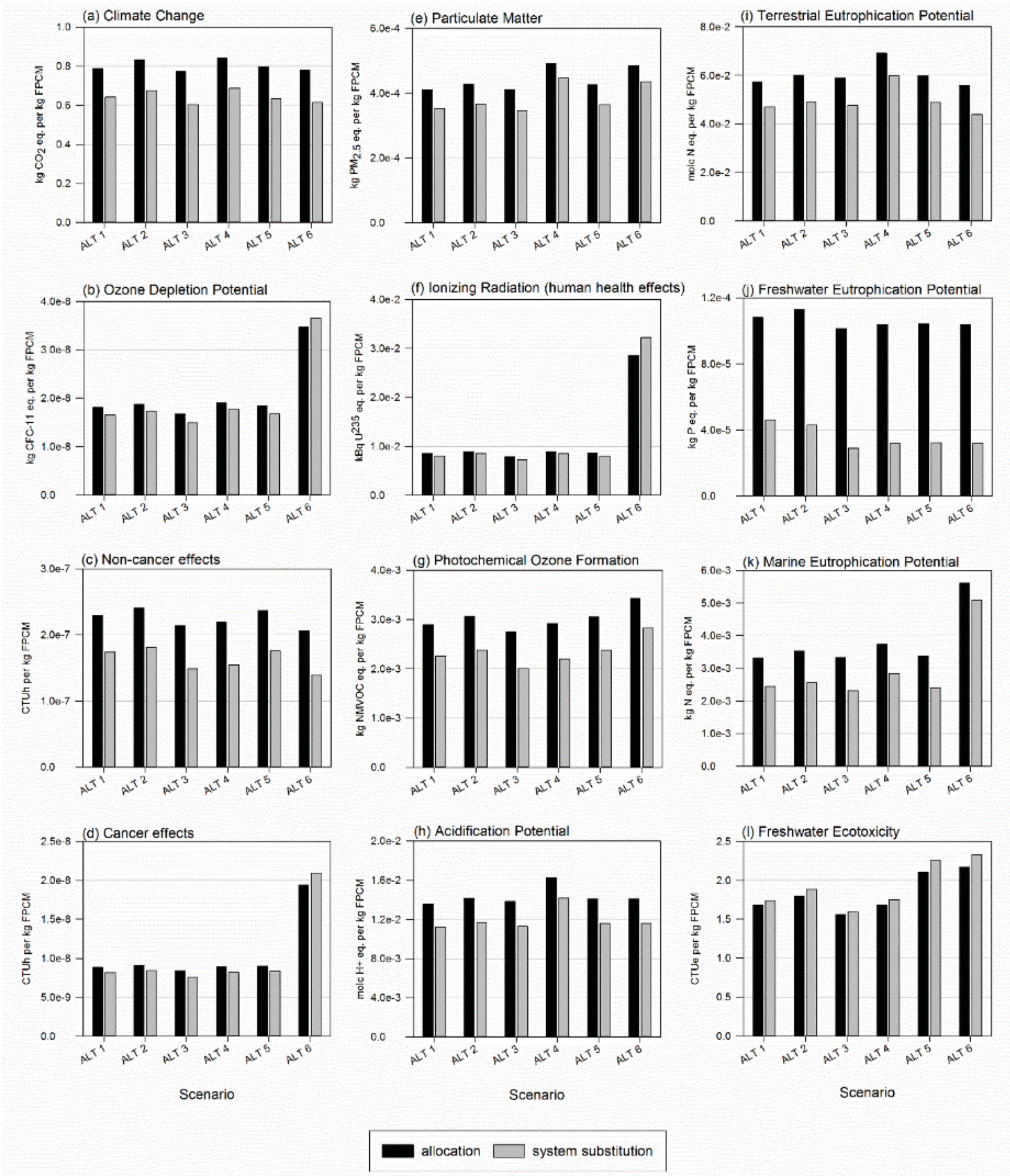

Figure 6.3 A comparison of environmental impacts derived from application of two different co-product handling methods for dairy co-products: (i) "allocation" refers to using biophysical relationship to partition environmental impacts between the milk and dairy meat, and (ii) "system substitution" refers to using system substitution to handle dairy co-products of dairy calves and dairy meat by substituting them with calves and beef originating from conventional beef systems, respectively.

6.4 Conclusions

Modeling the life cycle environmental impacts of prospective (future) product systems using scenario-based prospective ALCA can assist in selecting the most environmentally-friendly product, i.e. relatively lowest environmental impacts (Basset-Mens and van der Werf 2005; Basset-Mens et al. 2009; Dominguez-Ramos et al. 2010), and also in identifying the most effective mitigation or improvement option (scenario) to reduce the environmental impacts of a studied product system (Bauer et al. 2015; Leinonen et al. 2015; Sonesson et al. 2015).

The different prospective farm intensification methods generated different environmental profiles. Based on the results of the present study, it can be concluded that increased animal productivity is one of the most promising options for pasture-based dairy farming systems in the future as this option leads to reductions in all environmental indicators, on a per-kg of milk produced, when compared with increased stocking rate scenarios.

For the farm intensification methods involving increased stocking rate, the environmental benefits associated with improved pasture utilization efficiency were obvious. The use of additional N fertiliser to boost extra pasture production was associated with increased environmental impacts due to manufacturing and use of the additional N fertiliser. Increased use of local maize silage to intensify pasture-based dairy farms was another promising intensification option as almost all environmental indicators were improved when compared with increased use of a mixture of brought-in feeds, although measures to counteract the increased Ecotox indicator result need to be considered. The use of imported wheat grain was associated with higher environmental impacts compared with increased use of mixed brought-in feeds, due to the environmental impacts associated with wheat production systems and long-distance transport.

The results of the present study also highlighted the environmental hotspots for a range of likely future intensification options for pasture-based dairy farming systems. Therefore, it is important to seek effective measures to mitigate these hotspots in order to improve the environmental sustainability of future pasture-based milk production systems in New Zealand.

Using different co-product handling methods can result in alteration of environmental impacts of the life cycle of milk (Flysjö et al. 2011; Nguyen et al. 2013; Kiefer et al. 2015). In the present study, using system substitution to handle dairy co-products differently altered the magnitude of life cycle environmental impacts of milk when compared with those derived from using biophysical allocation method. This alteration was depended mainly on the magnitude of environmental impacts of the life cycle of conventional beef system in New Zealand, i.e. the displaced system. The higher values of the environmental impact categories for the life cycle of conventional beef systems resulted in the relatively lower environmental impacts for the life cycle of milk. However, system substitution method did not change the conclusion of the study since the relativity of the results remained unchanged when compared with allocation method.

Finally, estimation of future technologies is challenging and involves uncertainty (Lundie et al. 2004; Spielmann et al. 2005; Peters and Rowley 2009), especially for global product systems where many technologies/suppliers/processes involve different geographic origins (countries). Thus, there is a need for a widespread development and establishment of consensual prospective models and databases.

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Supplementary material E

E1. Selection of prospective dairy farming scenarios

Selection of prospective dairy farming scenarios were based on potential farm intensification methods. The selected farm intensification methods were according to recent farm practices and opinions of dairy experts. As a result, seven prospective dairy farming scenarios and one reference scenario were included, and are described in Table E1.

Table E1. Detailed description of selected prospective dairy farming scenarios in the Waikato region, New Zealand.

Scenarios	Description
REF	The reference scenario referred to an average pasture-based dairy farming system in the Waikato region for the production year 2010/11. Life cycle environmental impacts associated with 53 dairy farms in the Waikato region for the year 2010/11 have been characterized in Chapter 3. This scenario was used as a reference for making a relative comparison with the results derived from the BAU scenarios.
BAU	A dairy farming system in the business-as-usual (BAU) scenario was extrapolated using historical data associated with determining dairy farm traits (e.g. milk production per hectare, milk production per cow and stocking rate) in the Waikato region over the past 10 years (2005-2014) (DairyNZ 2014a). As a result, extrapolated milk production per hectare in 2025 was 30% greater than milk production in 2012, while milk production per cow was

	approximately 15% greater. These increases were assumed to be supported only by the use of additional brought-in feed.
ALT 1	Increased animal productivity (milk production per cow) in order to produce 20% more milk over the BAU scenario (to achieve a desired milk production target of 50% more milk relative to 2012). Extra feed demands were met by increased use of mixed brought-in feeds. This scenario represents farm intensification where cows were fed more in order to produce more milk/cow. Stocking rate was the same as the BAU scenario.
ALT 2	In this scenario, stocking rate was increased (more cows per hectare) in order to produce 20% more milk over the BAU scenario. Extra feed demands were met by increased use of mixed brought-in feeds. It was assumed that animal productivity (milk production per cow) was unchanged from the BAU scenario.
ALT 3	Expected milk production in this scenario was as in the ALT 2 scenario. However, extra feed demands were assumed to be derived from improved pasture utilization efficiency, i.e. cows harvested more feed from existing pastures. This intensification method is usually achieved through the use of skilled pasture and farm management practices (McCarthy et al. 2015).
ALT 4	This scenario was similar to the ALT 2 scenario, except that the additional feed required was derived from increased use of N fertilisers to boost extra pasture production. The amount of N fertiliser used was calculated based on the level of feed demand and the response rate of pasture to fertiliser N (Li et al. 2011).
ALT 5	This scenario was similar to the ALT 2 scenario, except that the additional feed required was derived from increased use of brought-in maize silage. Maize silage was assumed to be produced locally but off-farm.
ALT 6	This scenario was similar to the ALT 5 scenario, except that wheat grain rather than a mix of brought-in feed was used, and imported from a global market. The replacement of wheat grain for the mix of brought-in feed was based on the metabolisable energy concentration of feeds derived from Kolver (2000).

E2. Detailed flows and system boundary

In pasture-based dairy farming systems in New Zealand, while some farm inputs are locally produced, some are imported from overseas. For example, a large share of chemical fertilisers and pesticides are imported from the northern hemisphere, and palm kernel expeller comes from Malaysia. In Figure E1, detailed supplies and flows in the cradle-to-farm gate life cycle of pasture-based dairy farming system in the Waikato region, New Zealand are depicted.

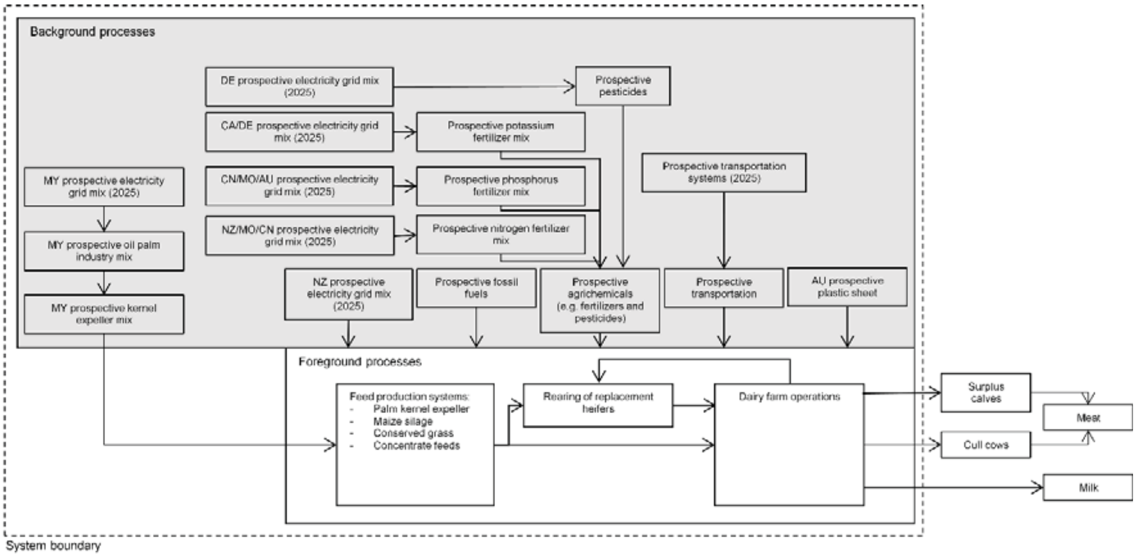

Figure E1 Simplified flows in pasture-based dairy farming system in the Waikato region, New Zealand (MY = Malaysia, DE = Germany, CA = Canada, CN = China, MO = Morocco, AU = Australia, NZ = New Zealand).

E3. Prospective electricity grid mix

In the present study, estimated environmental impacts associated with the electricity grid mix in individual countries were assessed using estimated shares of different electricity sources (e.g. coal-fired, geothermal, hydro and wind power plants) in 2025. The exception was for some countries where there was no estimation for 2025, and therefore the existing information was used. Prospective grid mixes for electricity are presented in Table E2, and multiple environmental impacts computed on a-per MJ of electricity grid mix basis are presented in Table E3.

Table E2 The estimated share (% of total) of electricity generation sources in 2025 (or as otherwise indicated) in the countries involved in the present study.

Electricity sources [a]	New Zealand [b]	China [c]	Canada (2020) [d]	Morocco (2012) [e]	Germany [f]	Malaysia (2020) [h]	Netherlands (2013) [i]
Coal	2.1	64.6	12.0	43.1	23.8	45.3	17.6
Oil			1.8	25.1	0.4	1.7	1.5
Natural gas	15.3	3.5	12.2	22.5	20.2	45.3	49.1
Hydropower	53.2	20.4	56.8	6.6	3.8	7.6	0.1
Geothermal	20.4				0.7		
Wind	8.6	6.89	3.3	2.6	25.6		3.4
Biomass			1.3		9.0		4.9
Nuclear		1.83	12.6				3.2
Photovoltaic		2.73	0.1		8.4		
Imported					8.1 [g]		20.3
Total	100	100	100	100	100	100	100

[a] Technologies associated with these electricity flows were assumed to be unchanged and obtained from the ecoinvent v3 database (ecoinvent Centre 2013), where 'high voltage electricity production, hard coal, allocation' was used as a proxy for hard coal-based system, 'high voltage electricity production, high voltage electricity production, oil, allocation' for oil-based system, 'high voltage electricity production, natural gas, combined cycle power plant, allocation' for natural gas-based system, high voltage electricity production, hydro, run-of-river, allocation' for hydropower system, high voltage electricity production, geothermal, allocation' for geothermal system, high voltage electricity production, wind, 1-3 MW turbine, offshore, allocation' for wind power system, 'high voltage electricity production, solid waste, allocation' for biomass-based system, 'high voltage electricity production, nuclear, pressure water reaction, allocation' for nuclear power system and 'high voltage electricity production, photovoltaic, 3 kWp slanted-roof installation, multi-Si, panel, mounted, allocation' for photovoltaic system; [b] data were derived from International Energy Agency (2011); [c] data were derived from Zou et al. (2015); [d] data were derived from International Energy Agency (2009); [e] data were derived from International Energy Agency (2015); [f] data were derived from Zimmermann et al. (2015); [g] assuming that imported electricity for use in Germany was equally imported from Austria, Czech Republic, Denmark, France, the Netherlands and Poland; data associated with electricity generation in these countries were obtained from the ecoinvent v3 database without modification (ecoinvent Centre 2013); [h] data derived from Tan et al. (2013); [i] flows of electricity grid mix derived from the ecoinvent v3 database (ecoinvent Centre 2013).

Table E3 Environmental impacts (on a per MJ basis) associated with the electricity grid mix for use in the prospective analysis (medium voltage).

Impact categories	Units	New Zealand	China	Canada	Morocco	Germany	Malaysia	Netherlands
CC	kg CO_2 eq.	3.03E-02	2.60E-01	6.01E-02	2.51E-01	1.35E-01	2.35E-01	1.77E-01
ODP	kg CFC-11 eq.	4.16E-09	1.06E-09	6.14E-09	1.03E-08	2.03E-08	1.56E-09	1.19E-08
Non-cancer	CTU_h	3.37E-09	2.33E-08	1.15E-08	2.37E-08	3.09E-08	3.36E-08	1.96E-08
Cancer	CTU_h	1.48E-09	4.21E-09	2.79E-09	6.02E-09	7.42E-09	9.44E-09	5.17E-09
PM	kg $PM_{2.5}$ eq.	9.97E-06	5.92E-04	3.33E-05	1.25E-04	2.79E-05	1.72E-04	2.24E-05
IR	kg U^{235} eq.	1.03E-03	6.32E-03	3.78E-02	4.66E-03	1.29E-02	1.72E-03	2.64E-02
POFP	kg NMVOC eq.	8.04E-05	9.87E-04	1.74E-04	8.06E-04	1.86E-04	4.36E-04	2.20E-04
AP	molc H^+ eq.	1.39E-04	2.96E-03	6.32E-04	1.96E-03	3.47E-04	1.17E-03	2.76E-04
TEP	molc N eq.	2.63E-04	3.49E-03	5.51E-04	2.74E-03	6.01E-04	1.50E-03	8.16E-04
FEP	kg P eq.	5.25E-06	3.49E-05	2.04E-05	6.68E-05	3.47E-05	1.12E-04	4.63E-05
MEP	kg N eq.	2.48E-05	3.25E-04	5.28E-05	2.62E-04	6.16E-05	1.57E-04	8.56E-05
Ecotox	CTU_e	1.50E-01	1.18E+00	5.98E-01	5.76E-01	2.60E+00	8.49E-01	5.29E-01

E4. Prospective fertilisers, lime and pesticides

In the present study, three main fertilisers (nitrogen [N], phosphorus [P] and potassium [K] assumed to be urea, single superphosphate [SSP] and potassium chloride [KCl], respectively, except for crop systems where diammonium phosphate [DAP] replaced SSP) and lime were used. The flows associated with manufacturing of these fertilisers and lime were derived from the ecoinvent v3 database (ecoinvent Centre 2013), and it was assumed that technologies for use to manufacture these fertilisers were unchanged. It should be noted that the prospective electricity grid mix was used to replace the existing electricity grid mix in the ecoinvent v3 database for the corresponding flows. Similarly, technologies associated with manufacturing of pesticides were assumed to be unchanged and imported from Germany. As a result, these pesticide flows were linked with the prospective German electricity grid mix. Multiple environmental impacts for these fertilisers, lime and pesticides are presented in Table E4.

Table E4 Environmental impacts associated with the manufacturing of chemical fertilisers, lime and pesticides (on a cradle-to-market perspective) used in the present study

Impact categories	Units	Urea (per kg N)	DAP (per kg P_2O_5)	DAP (per kg N)	SSP (per kg P_2O_5)	KCl (per kg K_2O)	Lime (per kg lime)	Glyphosate (per kg a.i.)	Pesticides (per kg a.i.)
CC	kg CO_2 eq.	3.15E+00	1.41E+00	2.76E+00	2.50E+00	5.89E-01	2.66E-02	1.12E+01	1.02E+01
ODP	kg CFC-11 eq.	1.48E-07	7.43E-08	1.46E-07	1.21E-07	3.09E-08	1.41E-09	2.37E-06	1.90E-05
Cancer	CTU_h	8.00E-07	1.06E-06	2.07E-06	1.89E-06	3.63E-07	5.19E-09	3.88E-06	3.69E-06
Non-cancer	CTU_h	7.79E-08	1.48E-07	2.90E-07	2.85E-07	4.10E-08	1.53E-09	5.82E-07	4.83E-07
PM	kg $PM_{2.5}$ eq.	5.72E-03	2.77E-03	5.44E-03	4.50E-03	3.15E-04	2.74E-05	9.40E-03	1.02E-02
IR	kg U^{235} eq.	1.35E-01	1.54E-01	3.02E-01	1.79E-01	4.66E-02	1.34E-03	2.11E+00	1.20E+00
POFP	kg NMVOC eq.	6.99E-03	6.51E-03	1.28E-02	1.13E-02	2.85E-03	1.72E-04	3.55E-02	4.13E-02
AP	molc H^+ eq.	2.96E-02	2.33E-02	4.58E-02	3.50E-02	3.67E-03	2.41E-04	8.14E-02	1.23E-01
TEP	molc N eq.	6.50E-02	2.35E-02	4.61E-02	3.73E-02	1.01E-02	6.64E-04	1.13E-01	1.27E-01
FEP	kg P eq.	4.77E-04	1.54E-03	3.02E-03	2.87E-03	2.19E-04	5.66E-06	1.53E-02	5.44E-03
MEP	kg N eq.	2.30E-03	1.91E-03	3.74E-03	3.55E-03	9.31E-04	5.58E-05	1.31E-02	2.14E-02
Ecotox	CTU_e	2.18E+01	2.59E+01	5.09E+01	4.73E+01	9.86E+00	1.30E-01	1.12E+02	1.48E+02

E5. Response of pasture to nitrogen fertiliser

In order to estimate amounts of nitrogen fertiliser (urea) required to boost pasture growth, a model associated with the response (i.e. kg of dry matter production per kg of urea nitrogen applied) of pasture to nitrogen fertiliser applied in the Waikato region was used. The model of Li et al. (2011) was applied and pasture nitrogen response rates are presented in Table E5.

Table E5 Response of pasture production (on a dry matter (DM) basis) to different application rates of nitrogen fertiliser.

Nitrogen fertiliser rate (kg N per ha per year)	Pasture response (kg DM/kg N)
111-230	16.8
231-363	16.6
364-502	15.2
503-627	12.3

Data based on model of Li et al. (2011)

E6. Sensitivity analyses

In sensitivity analyses, the influential effects of the choices associated with nitrogen fertiliser (urea) were tested. Three options included: Option (I) urea that was manufactured using the partial oxidation pathway in China, Option (II) urea that was manufactured by the steam reforming pathway in The Netherlands, and (III) urea based on Option (II) but assuming that energy use efficiency in liquid ammonia production was increased by 25%, and the manufacturing took place in New Zealand using electricity from wind power plants. These flows were linked with prospective electricity grid mix from the corresponding countries. In addition, some significant flows (e.g. heavy fuel oil for Option I and high pressure natural gas for Options II and III), were modified by assuming that these products were produced in those corresponding countries, as opposed to derive from global market. Multiple environmental impacts for these urea fertilisers (without their transportation to New Zealand) are presented in Table E6.

Table E6 Environmental impacts (on a per kg urea-N basis) associated with nitrogen fertilisers manufactured by using different pathways. The impacts were based on a cradle-to-market perspective.

Impact categories	Units	Option I Partial oxidation CN scenario	Option II Steam reforming NL scenario [a]	Option III Improved efficiency NZ scenario [b]
CC	kg CO_2 eq.	4.56E+00	2.87E+00 (-37%)	2.02E+00 (-30%)
ODP	kg CFC-11 eq.	2.97E-07	2.22E-07 (25%)	8.91E-08 (-60%)
Cancer	CTU_h	8.94E-07	7.62E-07 (-15%)	7.20E-07 (-6%)
Non-cancer	CTU_h	8.33E-08	7.39E-08 (-11%)	6.55E-08 (-11%)
PM	kg $PM_{2.5}$ eq.	1.10E-02	4.63E-03 (-58%)	3.67E-03 (-21%)
IR	kg U^{235} eq.	2.72E-01	1.47E-01 (-46%)	8.90E-02 (-39%)
POFP	kg NMVOC eq.	1.13E-02	4.43E-03 (-61%)	3.91E-03 (-12%)
AP	molc H^+ eq.	4.65E-02	1.82E-02 (-61%)	1.71E-02 (-6%)
TEP	molc N eq.	7.39E-02	6.06E-02 (-18%)	4.53E-02 (-25%)
FEP	kg P eq.	4.87E-04	4.81E-04 (-1%)	3.95E-04 (-18%)
MEP	kg N eq.	3.11E-03	1.89E-03 (-39%)	1.40E-03 (-26%)
Ecotox	CTU_e	2.63E+01	2.04E+01 (-22%)	2.05E+01 (0%)

[a] Values in parentheses referred to changes (%) in environmental impacts of urea manufacturing using the steam reforming pathway in NL (Netherlands) against the partial oxidation pathway in CN (China); [b] Changes (%) in urea manufacturing using the steam reforming pathway with 25% improved efficiency in NZ (New Zealand) against the steam reforming pathway in the Netherlands (NL).

E7. Environmental impacts for the prospective scenarios

Table E7 presents absolute values for multiple environmental impacts for each prospective scenario. The values were also split into the life cycle stages in pasture-based dairy farming systems.

Table E7. Absolute environmental impact values associated with prospective pasture-based dairy farming systems in the Waikato region, New Zealand.

1. Climate Change

	kg CO_2 eq. per kg FPCM						Percent (%) of total					
	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	4.87E-01	1.13E-01	1.56E-01	3.43E-02	3.67E-02	8.26E-01	59	14	19	4	4	100
ALT 1	4.61E-01	1.01E-01	1.61E-01	3.05E-02	3.68E-02	7.90E-01	58	13	20	4	5	100
ALT 2	4.73E-01	1.13E-01	1.77E-01	2.97E-02	3.98E-02	8.33E-01	57	14	21	4	5	100
ALT 3	4.63E-01	1.13E-01	1.36E-01	2.97E-02	3.18E-02	7.73E-01	60	15	18	4	4	100
ALT 4	5.13E-01	1.13E-01	1.36E-01	4.64E-02	3.52E-02	8.43E-01	61	13	16	6	4	100
ALT 5	4.70E-01	1.13E-01	1.52E-01	2.97E-02	3.31E-02	7.97E-01	59	14	19	4	4	100
ALT 6	4.30E-01	1.13E-01	1.41E-01	2.97E-02	9.73E-02	8.11E-01	53	14	17	4	12	100

2. Ozone Depletion Potential

	kg CFC-11 eq. per kg FPCM						Percent (%) of total					
ODP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	2.52E-09	1.06E-09	2.29E-09	1.86E-09	2.39E-09	1.01E-08	25	11	23	18	24	100
ALT 1	2.59E-09	9.47E-10	2.36E-09	1.66E-09	2.40E-09	9.95E-09	26	10	24	17	24	100
ALT 2	2.52E-09	1.06E-09	2.60E-09	1.61E-09	2.59E-09	1.04E-08	24	10	25	16	25	100
ALT 3	2.52E-09	1.06E-09	2.02E-09	1.61E-09	2.07E-09	9.28E-09	27	11	22	17	22	100
ALT 4	2.52E-09	1.06E-09	2.02E-09	2.40E-09	2.28E-09	1.03E-08	24	10	20	23	22	100
ALT 5	2.52E-09	1.06E-09	3.08E-09	1.61E-09	2.16E-09	1.04E-08	24	10	30	15	21	100
ALT 6	2.52E-09	1.06E-09	6.86E-09	1.61E-09	6.20E-09	1.82E-08	14	6	38	9	34	100

3. Human Health Toxicity-non-cancer effect

	CTUh per kg FPCM						Percent (%) of total					
Noncancer	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	9.18E-08	3.60E-08	9.96E-08	1.55E-08	4.20E-09	2.47E-07	37	15	40	6	2	100
ALT 1	8.19E-08	3.20E-08	1.03E-07	1.38E-08	4.24E-09	2.35E-07	35	14	44	6	2	100
ALT 2	7.97E-08	3.60E-08	1.13E-07	1.35E-08	4.58E-09	2.47E-07	32	15	46	5	2	100
ALT 3	7.97E-08	3.60E-08	8.73E-08	1.35E-08	3.65E-09	2.20E-07	36	16	40	6	2	100
ALT 4	8.11E-08	3.60E-08	8.73E-08	1.77E-08	3.95E-09	2.26E-07	36	16	39	8	2	100
ALT 5	7.97E-08	3.60E-08	1.10E-07	1.35E-08	3.92E-09	2.43E-07	33	15	45	6	2	100
ALT 6	7.97E-08	3.60E-08	1.66E-07	1.35E-08	8.68E-09	3.03E-07	26	12	55	4	3	100

4. Human Health Toxicity-cancer effect

	CTUh per kg FPCM						Percent (%)of total					
Cancer	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	3.34E-09	9.26E-10	2.03E-09	2.04E-09	1.09E-09	9.42E-09	35	10	22	22	12	100
ALT 1	3.02E-09	8.25E-10	2.09E-09	1.82E-09	1.10E-09	8.85E-09	34	9	24	21	12	100
ALT 2	2.94E-09	9.26E-10	2.30E-09	1.77E-09	1.18E-09	9.12E-09	32	10	25	19	13	100
ALT 3	2.94E-09	9.26E-10	1.79E-09	1.77E-09	9.45E-10	8.36E-09	35	11	21	21	11	100
ALT 4	2.95E-09	9.26E-10	1.79E-09	2.18E-09	1.03E-09	8.88E-09	33	10	20	25	12	100
ALT 5	2.94E-09	9.26E-10	2.37E-09	1.77E-09	1.00E-09	9.01E-09	33	10	26	20	11	100
ALT 6	2.94E-09	9.26E-10	1.29E-08	1.77E-09	2.48E-09	2.10E-08	14	4	61	8	12	100

5. Particulate Matter

	kg $PM_{2.5}$ eq. per kg FPCM						Percent (%) of total					
PM	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	2.02E-04	7.09E-05	7.19E-05	6.00E-05	3.05E-05	4.36E-04	46	16	16	14	7	100
ALT 1	1.85E-04	6.32E-05	7.41E-05	5.35E-05	3.06E-05	4.07E-04	46	16	18	13	8	100
ALT 2	1.86E-04	7.09E-05	8.16E-05	5.20E-05	3.30E-05	4.24E-04	44	17	19	12	8	100
ALT 3	1.95E-04	7.09E-05	6.32E-05	5.20E-05	2.64E-05	4.07E-04	48	17	16	13	6	100
ALT 4	2.43E-04	7.09E-05	6.32E-05	8.24E-05	2.97E-05	4.89E-04	50	15	13	17	6	100
ALT 5	1.82E-04	7.09E-05	9.04E-05	5.20E-05	2.70E-05	4.22E-04	43	17	21	12	6	100
ALT 6	1.76E-04	7.09E-05	2.11E-04	5.20E-05	9.43E-05	6.04E-04	29	12	35	9	16	100

6. Ionizing Radiation-human health effects

	kBq U^{235} eq. per kg FPCM						Percent (%) of total					
IR	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	9.41E-04	8.20E-04	1.24E-03	1.89E-03	3.44E-03	8.33E-03	11	10	15	23	41	100
ALT 1	9.67E-04	7.30E-04	1.28E-03	1.68E-03	3.45E-03	8.11E-03	12	9	16	21	43	100
ALT 2	9.41E-04	8.20E-04	1.41E-03	1.64E-03	3.73E-03	8.53E-03	11	10	17	19	44	100
ALT 3	9.41E-04	8.20E-04	1.10E-03	1.64E-03	2.98E-03	7.47E-03	13	11	15	22	40	100
ALT 4	9.41E-04	8.20E-04	1.10E-03	2.35E-03	3.31E-03	8.51E-03	11	10	13	28	39	100
ALT 5	9.41E-04	8.20E-04	1.62E-03	1.64E-03	3.08E-03	8.10E-03	12	10	20	20	38	100
ALT 6	9.41E-04	8.20E-04	1.58E-02	1.64E-03	9.38E-03	2.86E-02	3	3	55	6	33	100

7. Photochemical Ozone Formation Potential

	kg NMVOC eq. per kg FPCM						Percent (%) of total					
POFP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	1.77E-03	3.34E-04	3.91E-04	1.09E-04	4.05E-04	3.01E-03	59	11	13	4	13	100
ALT 1	1.71E-03	2.98E-04	4.03E-04	9.74E-05	4.07E-04	2.92E-03	59	10	14	3	14	100
ALT 2	1.78E-03	3.35E-04	4.44E-04	9.47E-05	4.39E-04	3.09E-03	58	11	14	3	14	100
ALT 3	1.65E-03	3.35E-04	3.42E-04	9.47E-05	3.51E-04	2.77E-03	60	12	12	3	13	100
ALT 4	1.74E-03	3.35E-04	3.42E-04	1.32E-04	3.95E-04	2.94E-03	59	11	12	4	13	100
ALT 5	1.75E-03	3.35E-04	5.45E-04	9.47E-05	3.57E-04	3.08E-03	57	11	18	3	12	100
ALT 6	1.36E-03	3.35E-04	5.63E-04	9.47E-05	1.27E-03	3.62E-03	38	9	16	3	35	100

8. Acidification Potential

	molc H^+ eq. per kg FPCM						Percent (%) of total					
AP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	8.61E-03	2.64E-03	2.31E-03	3.76E-04	6.37E-04	1.46E-02	59	18	16	3	4	100
ALT 1	7.80E-03	2.35E-03	2.38E-03	3.35E-04	6.38E-04	1.35E-02	58	17	18	2	5	100
ALT 2	7.88E-03	2.64E-03	2.62E-03	3.25E-04	6.88E-04	1.41E-02	56	19	19	2	5	100
ALT 3	8.26E-03	2.64E-03	2.02E-03	3.25E-04	5.52E-04	1.38E-02	60	19	15	2	4	100
ALT 4	1.04E-02	2.64E-03	2.02E-03	4.83E-04	6.24E-04	1.62E-02	64	16	12	3	4	100
ALT 5	7.68E-03	2.64E-03	2.84E-03	3.25E-04	5.57E-04	1.40E-02	55	19	20	2	4	100
ALT 6	7.40E-03	2.64E-03	1.95E-03	3.25E-04	2.08E-03	1.44E-02	51	18	14	2	14	100

9. Terrestrial Eutrophication Potential

	molc N eq. per kg FPCM						Percent (%) of total					
TEP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	3.85E-02	1.15E-02	1.00E-02	6.30E-04	1.47E-03	6.21E-02	62	19	16	1	2	100
ALT 1	3.49E-02	1.02E-02	1.03E-02	5.61E-04	1.47E-03	5.75E-02	61	18	18	1	3	100
ALT 2	3.52E-02	1.15E-02	1.14E-02	5.46E-04	1.59E-03	6.02E-02	58	19	19	1	3	100
ALT 3	3.69E-02	1.15E-02	8.79E-03	5.46E-04	1.27E-03	5.90E-02	63	19	15	1	2	100
ALT 4	4.66E-02	1.15E-02	8.79E-03	8.91E-04	1.43E-03	6.92E-02	67	17	13	1	2	100
ALT 5	3.43E-02	1.15E-02	1.24E-02	5.46E-04	1.29E-03	6.01E-02	57	19	21	1	2	100
ALT 6	3.31E-02	1.15E-02	7.00E-03	5.46E-04	4.65E-03	5.68E-02	58	20	12	1	8	100

10. Freshwater Eutrophication Potential

	kg P eq. per kg FPCM						Percent (%) of total					
FEP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	3.24E-05	1.64E-05	4.14E-05	1.80E-05	3.72E-06	1.12E-04	29	15	37	16	3	100
ALT 1	3.30E-05	1.46E-05	4.27E-05	1.61E-05	3.74E-06	1.10E-04	30	13	39	15	3	100
ALT 2	3.21E-05	1.64E-05	4.70E-05	1.56E-05	4.04E-06	1.15E-04	28	14	41	14	4	100
ALT 3	3.21E-05	1.64E-05	3.61E-05	1.56E-05	3.23E-06	1.03E-04	31	16	35	15	3	100
ALT 4	3.21E-05	1.64E-05	3.61E-05	1.82E-05	3.58E-06	1.06E-04	30	15	34	17	3	100
ALT 5	3.21E-05	1.64E-05	3.90E-05	1.56E-05	3.36E-06	1.06E-04	30	15	37	15	3	100
ALT 6	3.21E-05	1.64E-05	4.70E-05	1.56E-05	1.01E-05	1.21E-04	26	14	39	13	8	100

11. Marine Eutrophication Potential

	kg N eq. per kg FPCM						Percent (%) of total					
MEP	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	2.09E-03	3.42E-04	8.46E-04	3.53E-05	1.32E-04	3.44E-03	61	10	25	1	4	100
ALT 1	1.95E-03	3.05E-04	8.73E-04	3.14E-05	1.32E-04	3.29E-03	59	9	27	1	4	100
ALT 2	2.02E-03	3.42E-04	9.61E-04	3.06E-05	1.43E-04	3.50E-03	58	10	27	1	4	100
ALT 3	2.07E-03	3.42E-04	7.42E-04	3.06E-05	1.14E-04	3.30E-03	63	10	23	1	3	100
ALT 4	2.46E-03	3.42E-04	7.42E-04	4.28E-05	1.29E-04	3.71E-03	66	9	20	1	3	100
ALT 5	1.97E-03	3.42E-04	9.01E-04	3.06E-05	1.16E-04	3.36E-03	59	10	27	1	3	100
ALT 6	1.85E-03	3.42E-04	2.65E-03	3.06E-05	4.18E-04	5.28E-03	35	6	50	1	8	100

12. Ecotoxicity for Aquatic Freshwater

	CTUe per kg FPCM						Percent (%) of total					
Ecotox	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total	Onfarm	Replacement	Feeds	Agrichemicals	Transport	Total
BAU	1.17E-01	2.73E-01	7.87E-01	4.02E-01	1.10E-01	1.69E+00	7	16	47	24	6	100
ALT 1	1.09E-01	2.44E-01	8.12E-01	3.58E-01	1.11E-01	1.63E+00	7	15	50	22	7	100
ALT 2	1.06E-01	2.74E-01	8.94E-01	3.48E-01	1.20E-01	1.74E+00	6	16	51	20	7	100
ALT 3	1.06E-01	2.74E-01	6.85E-01	3.48E-01	9.53E-02	1.51E+00	7	18	45	23	6	100
ALT 4	1.06E-01	2.74E-01	6.85E-01	4.64E-01	1.02E-01	1.63E+00	6	17	42	28	6	100
ALT 5	1.06E-01	2.74E-01	1.22E+00	3.48E-01	1.03E-01	2.05E+00	5	13	60	17	5	100
ALT 6	1.06E-01	2.74E-01	1.26E+00	3.48E-01	2.09E-01	2.19E+00	5	12	57	16	10	100

E8. Environmental impacts for the reference scenario

In the present study, multiple environmental impacts associated with the average dairy farming system in the Waikato for the production year 2010/11 were used as a reference scenario. Multiple environmental impacts for the reference scenario are presented in Table E8. These data were derived from Chapter 3.

Table E8 Means, standard deviations (SD) and 95% confidence intervals (95% CI) of environmental impacts (per kg FPCM) of 53 dairy farms in the Waikato region, New Zealand for the year 2010/11.

Impact categories	Units	Mean	SD	95% CI	
				Lower limit	Upper limit
CC	kg CO_2 eq.	0.80E+00	0.08E+00	0.78E+00	0.82E+00
ODP	kg CFC-11 eq.	1.02E-08	0.17E-08	0.97E-08	1.06E-08
Non-cancer	CTU_h	2.60E-07	0.95E-07	2.33E-07	2.86E-07
Cancer	CTU_h	1.00E-08	0.40E-08	0.89E-08	1.11E-08
PM	kg $PM_{2.5}$ eq.	4.60E-04	0.74E-04	4.40E-04	4.80E-04
IR	kg U^{235} eq.	1.06E-02	0.29E-02	0.98E-02	1.14E-02
POFP	kg NMVOC eq.	2.58E-03	0.28E-03	2.50E-03	2.66E-03
AP	molc H^+ eq.	1.53E-02	0.19E-02	1.48E-02	1.59E-02
TEP	molc N eq.	6.55E-02	0.80E-02	6.33E-02	6.77E-02
FEP	kg P eq.	0.96E-04	0.19E-04	0.91E-04	1.02E-04
MEP	kg N eq.	2.67E-03	0.43E-03	2.55E-03	2.79E-03
Ecotox	CTU_e	1.23E+00	0.34E+00	1.14E+00	1.32E+00

E9. Results of sensitivity analyses

The influential effects of the source of nitrogen fertiliser (urea) on the final LCA results in the life cycle of dairy farming systems can be significant. Urea that was manufactured through two different pathways (partial oxidation and steam reforming pathways) was tested for its influence on the environmental impacts of pasture-based dairy farming systems. In general, net environmental impacts associated with the partial oxidation pathway were higher than those in the steam reforming pathways (see Table E6). The analysis was carried out in the ALT 4 scenario where extra urea was used to produce more pasture production and subsequently increased milk production per hectare. The results are presented in Table E9.

Table E9 Comparison of environmental impacts (on a per kg FPCM basis) in the ALT 4 scenario for different sources of urea fertiliser. The ALT 4 scenario refers to the prospective dairy farming system associated with increased stocking rate coupled with increased use of nitrogen fertiliser to boost pasture production.

Impact categories	Units	partial oxidation CN scenario	steam reforming NL scenario	Improved efficiency NZ scenario [b]
CC	kg CO_2 eq.	0.81E00	7.90E-01 (-3%)	0.77E00 (-3%)
ODP	kg CFC-11 eq.	1.19E-08	1.09E-08 (-8%)	8.80E-09 (-19%)
Noncancer	CTU_h	2.12E-07	2.11E-07 (-1%)	2.10E-07 (0%)
Cancer	CTU_h	8.65E-09	8.53E-09 (-1%)	8.27E-09 (-3%)
PM	kg $PM_{2.5}$ eq.	5.39E-04	4.57E-04 (-15%)	4.38E-04 (-4%)
IR	kg U^{235} eq.	1.01E-02	8.51E-03 (-15%)	7.10E-03 (-17%)
POFP	kg NMVOC eq.	2.81E-03	2.72E-03 (-3%)	2.62E-03 (-4%)
AP	molc H^+ eq.	1.57E-02	1.53E-02 (-2%)	1.52E-02 (-1%)
TEP	molc N eq.	6.62E-02	6.60E-02 (0%)	6.55E-02 (-1%)
FEP	kg P eq.	1.01E-04	1.01E-04 (0%)	9.90E-05 (-2%)
MEP	kg N eq.	3.59E-03	3.57E-03 (0%)	3.53E-03 (-1%)
Ecotox	CTU_e	1.55E00	1.48E+00 (-5%)	1.47E00 (-1%)

[a] Values in parentheses referred to changes (%) in environmental impacts of the life cycle of milk in ALT 4 when using urea derived from the steam reforming pathway in NL (the Netherlands) against that derived from the partial oxidation pathway in CN (China); [b] Changes (%) in environmental impacts of the life cycle of milk in ALT 4 when using urea derived from the steam reforming pathway with 25% improved efficiency in NZ (New Zealand) against that derived from the steam reforming pathway in the Netherlands (NL).

DRC 16

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STATEMENT OF CONTRIBUTION TO DOCTORAL THESIS CONTAINING PUBLICATIONS

(To appear at the end of each thesis chapter/section/appendix submitted as an article/paper or collected as an appendix at the end of the thesis)

We, the candidate and the candidate's Principal Supervisor, certify that all co-authors have consented to their work being included in the thesis and they have accepted the candidate's contribution as indicated below in the *Statement of Originality*.

Name of Candidate: Jeerasak Chobtang

Name/Title of Principal Supervisor: Sarah McLaren

Name of Published Research Output and full reference:

Chobtang, J., McLaren, S.J., Ledgard, S.F. and Donaghy, D.J. 2016. Environmental trade-offs associated with intensification methods in a pasture-based dairy system using prospective attributional Life Cycle Assessment. Journal of Cleaner Production (submitted)

In which Chapter is the Published Work: Chapter 6

Please indicate either:

- The percentage of the Published Work that was contributed by the candidate:

  and / or

- Describe the contribution that the candidate has made to the Published Work:

  The candidate was responsible for identifying feasible intensification options (scenarios) to increase milk productivity, developing prospective flows, performing environmental assessment, interpreting the results and writing the majority of the manuscript.

Jeerasak Chobtang

Candidate's Signature

12 May 2016

Date

Sarah McLaren

Principal Supervisor's signature

16 May 2016

Date

GRS Version 3– 16 September 2011

Chapter 7 Discussion on Methodological Choices in Life Cycle Assessment, Interpretation and Applicability of the Results

7.1 Introduction

Life Cycle Assessment (LCA) is usually used to assess the environmental impacts of a product or service over its life cycle; it can be used to support selection of environmentally-friendlier product system(s) and choice of environmental improvement (mitigation) options, strategies, decisions and policies (Finnveden and Moberg 2005; Hellweg and Milà i Canals 2014). In principle, LCA comprehensively accounts for use of resources and emissions released across the life cycle of a studied product system, starting from extraction of raw material through to manufacturing, distribution, use, and on to end-of-life (waste) management (Finnveden et al. 2009; EC-JRC-IES 2010; McManus and Taylor 2015). These resources and emissions are subsequently classified and characterized (transformed) into a range of environmental impact categories (indicators) based on their environmental cause-effect mechanisms (Finnveden et al. 2009; Hauschild et al. 2013; McManus et al. 2015). These environmental impact categories can be located either at the middle (i.e. midpoint level) or at the end (i.e. endpoint level) of environmental cause-effect chains (Finnveden et al. 2009; Hauschild et al. 2013). They are ultimately linked to impacts on the so-called 'areas of protection': (i) natural environment, (ii) natural resources and (iii) human health (EC-JRC-IES 2010).

As resources used and emissions released, throughout the life cycle of a studied product, are comprehensively accounted for, problems associated with burden-shifting between impact categories and between life cycle stages are revealed when comparing alternative product systems and/or considering improvement options (Finnveden et al. 2009; Hellweg and Milà i Canals 2014). In addition, a shift in environmental burdens between relevant businesses (sectors) can also be revealed when indirect effects are taken into account (Weidema et al. 2009; Brandão et al. 2014). This type of analysis can, therefore, assist users (e.g. decision-makers and system strategic planners) to prioritise the most relevant and feasible a preferred product system and/or improvement option(s) from a range of alternative product systems.

Even though LCA has been developed over the past few decades (McManus and Taylor 2015), a range of issues remain unresolved (Reap et al. 2008; Wardenaar et al. 2012). In particular, there is heated debate about whether LCA is truly effective to support policy- and decision-making (Anex and Lifset 2014; Brandão et al. 2014; Dale and Kim 2014;

Hertwich 2014; Plevin et al. 2014; Suh and Yang 2014). Additionally, issues associated with data to be used, e.g. average or marginal (Mathiesen et al. 2009; Ekvall et al. 2016), and methods to handle co-products (Weidema and Schmidt 2010; Wardenaar et al. 2012) in decision-supporting LCA studies, remain controversial. Recently, the potential of LCA to generate information associated with absolute environmental indicators, as opposed to relative environmental indicators, has received attention in the sustainable production and consumption research communities (Sandin et al. 2015; Frischknecht et al. 2016).

In the previous chapters, the environmental performance of different farm intensification options for future pasture-based dairy farming systems in the Waikato region was assessed. However, some relevant issues, that transcend the specific research reported in each of the individual chapters, were not addressed. Therefore, this chapter addresses a number of methodology-related and interpretation-related aspects not yet addressed in the previous chapters (Section 7.2), the applicability of LCA to support decision-making (Section 7.3), and the potential of LCA to generate absolute environmental information as opposed to relative environmental information (Section 7.4).

7.2 Methodology-related and interpretation-related aspects

In this section, issues that are potentially important in the relationship between LCA methodology and interpretation of LCA results are identified and discussed in depth. They are: the influence of different data sources for background systems (Section 7.2.1); the effect of different functional units (Section 7.2.2); the influence of different co-product handling methods (Section 7.2.3); the variability in environmental indicators relative to different dairy farming practices and potential improvement options (Section 7.2.4); and the environmental benefits and disadvantages of intensified pasture-based dairy systems in the future relative to the current dairy system (Section 7.2.5).

7.2.1 Influence of different data sources for background systems

Life Cycle Assessment is a data-intensive approach that involves collection of data along the entire life cycle from extraction of raw material through to end-of-life (waste) management (i.e. a cradle-to-grave perspective). Therefore the quality of data plays a critical role in the credibility and reliability of an LCA study (Weidema et al. 2013; Pelletier 2015).

In the present research, primary data associated with foreground systems[4] were usually derived from direct measurement whereas most data in background systems[5] were derived mainly from existing databases, e.g. ecoinvent (ecoinvent Centre 2013) and Agri-Footprint (Agri-Footprint 2014). In addition to inherent differences associated with different databases (Takano et al. 2014; Herrmann and Moltesen 2015; Speck et al. 2016), frequent updates of these individual databases due to progress in science and technology as well as improved accuracy of measurement or modelling, can result in changes in the magnitude of environmental emissions associated with processes (Steubing et al. 2016). In the present study, the ecoinvent database (ecoinvent Centre 2013) was used as the main source of data for background processes. Initially, the ecoinvent database version 3.0 was used. During this research, the ecoinvent database was updated to version 3.1. Therefore, in order to investigate the influence of the database update on the environmental impacts for the cradle-to-farm gate life cycle of milk production systems, the 53 individual dairy systems (farms) used for the study described in Chapter 3, were re-assessed using the ecoinvent database version 3.1. The results were compared with the original results in Chapter 3 where the ecoinvent database version 3.0 was used.

The differences between environmental indicator results per kg FPCM are shown in Table 7.1. For two impact categories (ODP and MEP), the results substantially increased (by 66% and 16%, respectively) whereas for five other impact categories (Cancer, Non-cancer, IR, FEP and Ecotox) the results decreased by 5 – 12%. However, the changes did not alter the dominating substances for these impact categories. For example, for ODP and MEP indicators, bromotrifluoro-methane (Halon 1301) and NO_3^- remained the dominating contributors, respectively. In contrast, the change in results for the CC, PM, POFP, AP and TEP indicators were insignificant (less than 3%). These

[4] The processes of a system that are specific to such a system. These processes are usually controlled directly by the producer of a studied product (EC-JRC-IES 2010).

[5] The processes of a system that are not under direct control of the producer of a studied product. These processes are generally market-averaged (EC-JRC-IES 2010).

changes were associated mainly with new and updated data in ecoinvent database version 3.1 (Moreno Ruiz et al. 2014; Wernet et al. 2016).

In summary, the change in some of the results underlined the importance of using the most up-to-date versions of LCA databases (assuming that these changes actually do improve the accuracy of the datasets).

Table 7.1 A comparison of environmental impacts per kg of fat- and protein-corrected milk (mean and standard deviation [SD]) of the cradle-to-farm gate life cycle of pasture-based dairy farming systems using different versions of the ecoinvent database. Impacts are expressed on a per-kg of fat- and protein-corrected milk, and averaged over 53 dairy systems (farms).

Impact categories	Units [a]	Ecoinvent v 3.0		Ecoinvent v 3.1		Difference (%)
		Mean (*x*)	SD	Mean (*y*)	SD	$(y\text{-}x)/x \times 100$
CC	kg CO_2 eq.	0.76E+00	0.08E+00	0.75E+00	0.07E+00	-1.3
ODP	kg CFC-11 eq.	1.02E-08	0.17E-08	1.69E-08	0.34E-08	+65.7
Cancer	CTU_h	1.00E-08	0.40E-08	0.88E-08	0.38E-08	-12.4
Non-cancer	CTU_h	2.60E-07	0.95E-07	2.48E-07	0.92E-07	-4.6
PM	kg $PM_{2.5}$ eq.	4.60E-04	0.74E-04	4.55E-04	0.71E-04	-1.1
IR	kBq U^{235} eq.	1.06E-02	0.29E-02	0.98E-02	0.25E-02	-7.5
POFP	kg NMVOC eq.	2.58E-03	0.28E-03	2.52E-03	0.24E-03	-2.3
AP	molc H^+ eq.	1.53E-02	0.19E-02	1.52E-02	0.19E-02	-0.7
TEP	molc N eq.	6.55E-02	0.80E-02	6.53E-02	0.79E-02	-0.3
FEP	kg P eq.	0.96E-04	0.19E-04	0.90E-04	0.15E-04	-6.7
MEP	kg N eq.	2.67E-03	0.43E-03	3.06E-03	0.50E-03	+14.6
Ecotox	CTU_e	1.23E+00	0.34E+00	1.09E+00	0.32E+00	-11.4

[a] CO_2 stands for Carbon dioxide, eq. = equivalent, CFC-11 = Trichlorofluoro-methane, CTU_h = Comparative toxic unit for humans, $PM_{2.5}$ = Particulate matter less than 2.5 μ in diameter, kBq = kilobecquerel, U^{235} = Uranium-235, NMVOC = Non-methane volatile organic compounds, molc = mole of charge, H^+ = Hydrogen ion, N = Nitrogen, P = Phosphorus and CTU_e = Comparative toxic unit for ecosystems.

7.2.2 Effect of different functional units

Agricultural systems provide a range of functions, including: (i) production of food, feed, biofuel and fibre, (ii) maintaining landscape quality and ecosystem services, and (iii) recreational uses (Ripoll-Bosch et al. 2013; van der Werf et al. 2014; Weiler et al. 2014). Food production is one of the most important functions of agricultural systems, including dairy farming systems. In relation to food production, there are several functional units that can be used to represent this function of agricultural systems; they include mass of agricultural product (e.g. kg of milk), area of agricultural land (e.g. hectare of farmland), and mass of nutritional value (e.g. kg of total product protein). The ISO 14044 standard (International Organization for Standardization 2006) emphasises that the functional unit should correspond to the goal and scope of an LCA study.

In this research, one kg of FPCM (International Dairy Federation 2010) was used as a functional unit. It was chosen because milk production is normally considered as the main function of dairy farming systems. However, based on this functional unit, some types of environmental information that support either area-based or nutrition-based environmental perspectives are not represented. In general, the area-based functional unit is usually used to support identification of an alternative option associated with land use (management) efficiency (Nemecek et al. 2011; Ribal et al. 2016; Wang et al. 2016) and may be used to support decision-making associated with alternative land use strategies. For the nutrition-based functional unit (e.g. protein and energy), it generally includes total product nutrients produced per hectare of farmland (e.g. total edible protein production per hectare, regardless of types of products produced, for instance milk and meat) and is usually used to assist in identification of a better product system (e.g. more edible protein production per hectare of farmland) with higher resource use efficiency (e.g. farm inputs such as fertilisers and brought-in feeds) (Heller et al. 2013; Dyer and Vergé 2015; Nemecek et al. 2016). Therefore, it is worth considering additional use of other functional units that complement the results per kg FPCM, in order to provide insights into environmental sustainability from different perspectives.

In order to investigate this aspect, the two contrasting levels of dairy farming intensification (high versus low) previously assessed in the study described in Chapter 4, were re-assessed using two alternative functional units: (i) one hectare of dairy farmland (to support an area-based perspective), and (ii) one kg of total product protein (milk protein + meat protein) (to support a whole farm nutrition-based perspective). For the former, emissions were calculated based on one hectare of effective dairy farmland, regardless of the amount of milk or dairy co-product production. For the latter, total milk and meat production per hectare of effective dairy farmland was converted to total product protein.

The relative changes in ranking of environmental indicators between the high and low levels of dairy farm intensification characterised based on the three different functional units are presented in Table 7.2. The ranking of an individual pair of results for the high and low intensification levels for each impact category was consistent when they were modelled based on a per-kg FPCM and a per-kg total product protein basis. However, the ranking for two impact categories (IR and FEP) was changed when they were characterised based on a per-hectare of dairy farmland basis. The results for these two indicators on a per-hectare basis were significantly higher for the high intensification level than for the low intensification level whereas for the other two functional units, they were similar for the high and low intensification levels. Thus, results for seven indicators were significantly higher for the high dairy intensification group then for the low dairy intensification group on a per-hectare basis, whereas only five indicators were significantly higher for the high intensification group when computed based on the other two functional units. Product-based functional units (e.g. kg FPCM and kg of total product protein) usually represent the main product(s) and function(s) of agricultural systems.

In some cases, a very high intensification system (high milk production per hectare) may have relatively low emissions per kg FPCM but total emissions per hectare can still be higher for the high intensification system compared with a lower intensification level (e.g. Battini et al. 2016). In particular, an area-based perspective can be important at a local level where there is a specific local resource or environmental constraint, e.g. a lake or river catchment with a nutrient loss limit or a maximum acceptable eutrophication level. For example, the results in Table 7.1 showed that there were no significant differences in the FEP indicator results between low and high levels of farm intensification when computed on a per-kg FPCM basis. In contrast, on a per-hectare basis, the high intensification dairy system had significantly higher FEP indicator result than the low intensification dairy system. There is a nutrient limit (i.e. nutrient losses per hectare of farmland are capped by laws or regulations) for freshwater bodies (or catchment) in the Waikato region. Therefore the preferred choice based on the additional information provided with the FEP per-hectare results is the low level of dairy farm intensification (despite the similar FEP indicator results per-kg FPCM in low and high levels of farm intensification).

In summary, different functional units can be used when assessing environmental performance of dairy farming systems, depending on the goals or objective of the study.

Table 7.2 A comparison of multiple life cycle environmental indicators of two contrasting levels of dairy farm intensification in the Waikato region for three functional units.

Impact categories	1 kg FPCM		Sig. [a]	1 ha of dairy farmland		Sig. [a]	1 kg of protein produced [b]		Sig. [a]
	Low	High		Low	High		Low	High	
CC	6.82E-01	8.02E-01	***	0.91E+04	1.49E+04	***	2.38E+01	2.81E+01	***
ODP	1.37E-08	1.92E-08	***	1.83E-04	3.55E-04	***	4.80E-07	6.73E-07	***
Cancer	9.77E-09	7.84E-09	ns	1.30E-04	1.46E-04	ns	3.41E-07	2.74E-07	ns
Non-cancer	2.63E-07	2.44E-07	ns	3.50E-03	4.56E-03	ns	9.16E-06	8.55E-06	ns
PM	3.90E-04	4.95E-04	***	5.20E+00	9.14E+00	***	1.37E-02	1.73E-02	***
IR	0.89E-02	1.00E-02	ns	1.19E+02	1.86E+02	***	3.12E-01	3.51E-01	ns
POFP	2.29E-03	2.73E-03	ns	3.08E+01	5.07E+01	***	8.01E-02	9.56E-02	***
AP	1.35E-02	1.62E-02	***	1.79E+02	3.00E+02	***	4.71E-01	5.69E-01	**
TEP	5.80E-02	6.92E-02	***	0.77E+03	1.28E+03	***	2.03E+00	2.42E+00	**
FEP	8.82E-05	9.51E-05	ns	1.18E+00	1.77E+00	***	3.08E-03	3.33E-03	ns
MEP	2.82E-03	3.33E-03	*	3.76E+01	6.16E+01	***	9.84E-02	1.17E-01	*
Ecotox	1.15E+00	1.03E+00	ns	1.53E+04	1.93E+04	ns	4.04E+01	3.59E+01	ns

[a] significant levels derived from a t-test comparison between the low and high intensification groups are *** = P<0.001, ** = P<0.01, * = P<0.05, and ns = non-significance (P>0.05); [b] the protein concentration in dairy meat was derived from Beef+LambNZ (2016). Note that the dairy systems were modelled using ecoinvent database version 3.1 and the comparative environmental impacts between high and low levels of farm intensification were statistically analysed using a group t-test comparison.

7.2.3 Influence of different co-product handling methods

In LCA, the choice of a co-product handling method is considered one of the most controversial issues (Reap et al. 2008; Wardenaar et al. 2012; Pelletier et al. 2015). This is due to the fact that there are different methods to handle co-products, and each method can generate different environmental information, and hence implications (Flysjö et al. 2011; Nguyen et al. 2013; Kiefer et al. 2015). As a result, using different co-product handling methods can lead to different conclusions in LCA studies (Luo et al. 2009; Wardenaar et al. 2012).

The ISO 14044 standard (International Organization for Standardization 2006) recommends using a hierarchical approach when choosing co-product handling methods, and emphasises that the co-product handling methods shall be consistent along the life cycle of a studied product. It states that, firstly, all co-products shall be handled as far as possible using system subdivision; this can be done by separating production systems into different subsystems. However, most production systems have a long list of intermediate systems along with a range of co-products, as well as having a number of processes that cannot be subdivided. Therefore, it is nearly impossible for this method to be used to address all co-products throughout the entire supply chain.

The ISO standard states that, secondly, co-products should be handled using system expansion or system substitution, where the co-product(s) is substituted by a corresponding product(s) that has similar functions and/or market segments (Ekvall and Weidema 2004; Weidema and Schmidt 2010). In practice, the emissions of to-be-displaced product system(s) are subtracted from those associated with the studied product system. However, identification of a to-be-displaced product system(s) through market mechanisms is not straightforward since market elasticity and/or substitution rates of products are rather complex, especially when such products are traded in global markets.

Finally, the ISO standard states that, in any case where system substitution cannot be applied, co-products should be handled using particular relationships between the co-products e.g. physical relationship (e.g. mass and energy) or (ii) economic relationship (i.e. economic revenues of the co-products).

For the dairy sector, the International Dairy Federation (2010) provides a generic guideline for co-product handling methods. For the cradle-to-farm gate milk production system, the guideline recommends using an economic allocation for system inflows (e.g. feed production systems), and a biophysical relationship for system outflows (e.g. milk and dairy meat). It should be noted that these guidelines are not in line with the ISO standards with respect to consistency of co-product handling methods.

For pasture-based dairy systems in New Zealand, apart from a number of co-products generated from system inflows (e.g. palm oil versus PKE and barley versus barley straw), dairy meat derived from culled cows and surplus calves (bobby calves), and live surplus calves for use as young stock in beef farming systems are the main dairy co-products. In this research, different methods to account for these two co-products were tested, and the results are presented in Chapter 6. It was found that the main challenge in following the ISO standards to handle dairy co-products was associated with: (i) identification of representative to-be-displaced product systems (e.g. conventional beef farming systems), and (ii) availability of relevant environmental datasets. For example, the market for beef products derived from conventional beef farming systems (e.g. conventional cow-calf systems) can be considered to have the same function and market segmentation as dairy meat (meat derived from culled cows and surplus bobby calves). However, extra dairy meat appearing in the market does not always result in reduced conventional beef production (Thomassen et al. 2008). In the Netherlands, Thomassen et al. (2008) reported that increased dairy beef in the Dutch market also affected the level of pork production. Additionally, Flysjö et al. (2011) considered that meat derived from New Zealand bobby calves was similar to and could potentially displace chicken meat rather than beef derived from conventional beef systems.

Moreover, identification of representative global beef systems can be very challenging since there are a number of beef exporters (i.e. companies or nations) around the world, and this beef can be produced from different farming systems/practices (e.g. intensive versus extensive systems, and pasture-based versus concentrate-based systems) (de Vries et al. 2015). Ekvall and Weidema (2004) provided a simple 5-step approach to identify a single marginal supplier for use in a CLCA study; however, this approach can be difficult and challenging when the market for a marginal product (i.e. beef) is at a global scale (i.e. internationally traded products). This is because marginal products can be derived from different countries and different production systems with different production costs. Marvuglia et al. (2013) and Vázquez-Rowe et al. (2013) recommended using global economic models (e.g. a Partial Equilibrium model) to identify marginal suppliers (products) when the market for a marginal product is at a global scale. However, this approach usually leads to increased uncertainty of assessment results since there are additional assumptions and limitations of global economic models (Zamagni et al. 2012; Dale and Kim 2014). In this research, identification of a marginal beef mix for use to handle the co-product dairy meat (i.e. assuming dairy meat was substituted beef derived from conventional beef systems) was simplified as a conventional beef production system in New Zealand and a proportional mix of exported beef derived from the three largest exporting nations in the world was tested in a sensitivity analysis; the results are presented in Chapter 5. The findings demonstrate that the environmental impacts per kg FPCM depended largely on identified marginal beef systems. For example, the CC indicator result per kg FPCM was substantially decreased (24% – 40%, depending on dairy scenarios) when the global beef mix was used compared with using New Zealand beef system. This was because the conventional beef system derived from a global mix had a substantially higher CC impact than that derived from New Zealand conventional beef systems.

Furthermore, to be consistent with the ISO standards, all other co-products in the studied system inflows (e.g. palm oil versus PKE) should be handled using system substitution. Identification of their to-be-displaced product systems (e.g. a mix of global vegetable oils to displace palm oil) can be very challenging.

In conclusion, this research confirmed that the choice of a co-product handling method can have a large influence on the LCA results. Allocation methods can be used when the environmental burdens of a studied system need to be isolated from the rest-of-the-world burdens. This method is usually used when identification of environmental hotspots is the focus of attention or when there is no interest in accounting for the consequential effects associated with co-products. In contrast, system substitution can be used when consequential effects associated with the life cycle of co-products are of interest. However, identification of to-be-substituted product systems can be challenged for both average and marginal suppliers, especially when the to-be-substituted product systems occur on a global scale. Therefore, in order to produce comprehensive and transparent results, a scenario analysis or sensitivity analysis to test (likely) alternative to-be-substituted product systems should be performed.

7.2.4 Relation between physical farm traits and environmental impacts and implications for improvement options

In dairy farming systems, information derived from correlation analysis can be used to identify farm practices that are linked with higher farm productivity and reduced environmental impacts (Thomassen et al. 2009; Battini et al. 2016; O’Brien et al. 2016). The results of correlation analysis presented in Chapter 4 highlighted significantly useful implications. For example, pasture consumption per hectare was significantly negatively correlated with most environmental indicators per kg FPCM, implying that interventions that can result in increased pasture intake (i.e. improved pasture utilisation efficiency) can reduce most environmental impacts per kg FPCM. In contrast, the increased use of N and P fertilisers was significantly positively correlated with most impact categories, implying that increased use of these fertilisers can result in increased environmental indicators per kg FPCM.

7.2.5 Environmental performance of current and future dairy systems

In general, environmental assessment (e.g. using ALCA) of existing dairy systems can identify environmental hotspots that can become a focus for improvement or mitigation options. Equally, estimation of environmental impacts and environmental hotspots of future dairy systems can support future environmental planning (strategy) development and decision-making, as well as facilitate research directions to further improve the future environmental performance of dairy systems.

In the present research, both existing (current) and different future dairy farming scenarios were assessed. The 'business-as-usual' growth was used to develop models of dairy systems in the short-term to mid-term future (e.g. in 2025). The models included progress or development in some relevant technologies. The results are presented in Chapter 6.

In order to investigate the relevance of accounting for future development of technologies, the potential environmental impacts for the cradle-to-farm gate life cycle of existing dairy systems (see Chapter 3) and future 'business-as-usual' dairy system scenarios (see Chapter 6) were compared. These two (average) dairy farming scenarios were re-assessed using the updated ecoinvent database (version 3.1) for their background systems. As shown in Table 7.3, the results per kg FPCM for six environmental indicators (CC, Cancer, POFP, FEP, MEP and Ecotox) for the future dairy system were increased when compared with the current dairy system, while five environmental indicators (ODP, PM, IR, AP and TEP) were decreased. The Non-cancer indicator result remained unchanged.

Table 7.3 Comparative environmental performance per kg of fat- and protein-corrected milk between the existing (current) and future dairy farming scenarios in the Waikato region, New Zealand.

Impacts	Unit	Dairy farming systems [a]		Difference (%) $(y-x)/x \times 100$
		Current (x) [b]	Future (y) [c]	
CC	kg CO_2 eq.	0.75E+00	0.83E+00	+10.7
ODP	kg CFC-11 eq.	1.69E-08	1.01E-08	-40.2
Cancer	CTU_h	0.88E-08	0.94E-08	+6.8
Non-cancer	CTU_h	2.48E-07	2.47E-07	-0.4
PM	kg $PM_{2.5}$ eq.	4.55E-04	4.36E-04	-4.2
IR	kBq U^{235} eq.	0.98E-02	0.83E-02	-15.3
POFP	kg NMVOC eq.	2.52E-03	3.01E-03	+19.4
AP	molc H^+ eq.	1.52E-02	1.46E-02	-3.9
TEP	molc N eq.	6.53E-02	6.21E-02	-4.9
FEP	kg P eq.	0.90E-04	1.12E-04	+24.4
MEP	kg N eq.	3.06E-03	3.44E-03	+12.4
Ecotox	CTU_e	1.09E+00	1.69E+00	+55.0

[a] The current and future dairy systems were re-assessed using the ecoinvent database version 3.1 for the background systems; [b] environmental impacts were averaged over 53 dairy systems (see Chapter 3); [c] this farm system was equivalent to the 'business-as-usual' scenario, as presented in Chapter 6.

The increase in some environmental impacts was mainly associated with increased use of brought-in feed to support increased stocking rate in future dairy farming systems. In contrast, the decrease in the other environmental indicators resulted mainly from: (i) increased animal productivity (~15%), and (ii) progress of technologies, especially electricity grid mix that is expected to have a larger share of environmentally-friendly technologies, e.g. hydro, wind power and renewable-related power plants.

In summary, in the absence of any additional interventions, if the growth rate of dairy farming systems in the Waikato continues as usual (estimated using historical data over the past ten years), there will be a considerable increase in some environmental impacts per kg FPCM. This implies that innovative system management practices are needed in order to counteract the increase in environmental impacts. This is particularly relevant for the production of off-farm feeds if they continue to be the main source of future dairy intensification.

7.3 Applicability of LCA approaches to dairy systems

Even though LCA modelling approaches can be seen as being along a continuous spectrum (Suh and Yang 2014), two distinctive LCA approaches have been recognised: ALCA and CLCA (Weidema et al. 2009; Plevin et al. 2014). First, ALCA has been defined as "*an attempt to provide information on what portion of global burdens can be associated with the life cycle of a studied product*" (UNEP/SETAC 2011 p. 47), and for CLCA as "*an attempt to provide information on the environmental burdens that occur, directly or indirectly, as a consequence of a decision (usually represented by changes in demand for a product)*" (UNEP/SETAC 2011 p. 47). Therefore, these two approaches generate different sets of environmental information and have different implications (Tillman 2000; UNEP/SETAC 2011).

Generally, ALCA is recommended to assess the environmental impacts of the life cycle of a studied product based on a key assumption that the studied system is static, i.e. a *status quo* assumption, meaning that all flows within a studied system are assumed to be fixed. Moreover, ALCA generates environmental information that is used to identify environmental hotspots, and hence opportunities for mitigation options or improvement options (Pelletier et al. 2015). In contrast, CLCA generates information associated with net environmental impacts resulting from an action, or a change in demand for a studied product, based on a *ceteris paribus* assumption, meaning that change within a studied system is possible, whereas for other related systems, changes are not allowed and these related systems are assumed to be constant. In CLCA, the net environmental impacts are generally presented in a relative form (relative to the system without such an action) (Weidema et al. 2009). Furthermore, CLCA can account for indirect effects (e.g. ILUC and secondary consequences, such as rebound effects) arising as a consequence of such an action. However, since CLCA requires marginal flows (a mix of flows where constrained flows are excluded) which do not represent 'overall-average' flows (i.e. a mix of flows that represented both constrained and unconstrained suppliers), CLCA results are not representative of 'overall-average' environmental impacts.

7.3.1 Proposed decision-context situations

There have been attempts to define and link different decision-context situations with different types of LCA modelling approaches (Ekvall et al. 2005; EC-JRC-IES 2010). For example, Ekvall et al. (2005) classified LCA into two types based on normative ethics: retrospective and prospective. Retrospective LCA provides information on the environmental impacts associated with a product; prospective LCA provides information related to the environmental consequences of an action. Curran et al. (2005) describe two LCI approaches for use to address different questions: attributional LCI and consequential LCI. The attributional LCI aims to attribute environmental burdens to a product being produced in the economy, while the consequential LCI can be used to assess environmental impacts resulting from implementing decisions.

The ILCD Handbook (EC-JRC-IES 2010) provides a guideline for selecting appropriate LCA modelling approaches to support different decision-contexts and differentiates three decision-context situations: Situations A, B and C (Table 7.4).

Table 7.4 The ILCD Handbook's guideline to determine different decision-context situations provided.

Decision support?		Kind of process-changes in background system/other systems	
	Yes	None or small-scale	Large-scale
		Situation A: Micro-level decision support	Situation B: Meso/macro-level decision support
	No	Situation C: C1 when existing interactions with other systems are included, C2 when no existing interactions with other systems are accounted for.	

Source: The ILCD Handbook (EC-JRC-IES 2010 p. 38).

As seen in Table 7.4, the ILCD Handbook defines two questions that need to be addressed in order to select the appropriate LCA approach. The first question is "*whether a decision is to be supported by an LCA study*". In the case where a decision is to be supported by an LCA study (i.e. Situations A or B), it is important to account for any consequences resulting from such a decision. In contrast, for an LCA study that is not intended to support decision-making (Situation C), no consequence needs to be accounted for.

The second question is "*whether the consequence of such a decision is small or large*". The consequence refers to the size of the effect of such a decision on background systems or other relevant systems. In the case where the decision has no effects or only small effects on the background systems or other relevant systems (non-structural change), this study will fall into Situation A (micro-level decision support). However, in the case where the scale of the affected systems is large (structural change), this is referred to Situation B (meso/macro-level decision support). In order to further clarify, a non-structural change generally refers to a change that needs no newly-installed equipment, meaning that such an action or decision or change can be supported by existing equipment capacity. In contrast, a large scale consequence refers to where at least part of the technology or equipment in the background systems or other systems in the economy needs to be newly-installed, meaning that existing equipment capacity cannot support such an action (decision).

For Situation A (micro-level decision support), ALCA is recommended to assess the environmental impacts of the life cycle of a studied product. In addition, all flows should represent an average market (consumption) mix. All valuable co-products occurring along the life cycle of a product system are recommended to be handled using system expansion/substitution. It is also recommended that the to-be-substituted product system should represent an average market (consumption) mix that has the same function and/or market segment to the corresponding co-product. In the case where system substitution is not available, allocation based on physical properties (first option) or economic values (second option) of the co-products is recommended. Note that this decision-context is usually associated with product-related questions.

For Situation B (meso/macro level decision support), ALCA is recommended, as in Situation A, to assess environmental impacts of the life cycle of a studied product with an average market (consumption) mix to represent inventory flows. The exception is that the inventory flows (in the background system) that are largely affected by such an action/decision/change should be modelled using a mix of the long-term marginal flows (CLCA) and handling co-products using system substitution. Note that this decision-context is usually associated with strategy-or policy-related questions.

For Situation C (accounting), ALCA is recommended, as in Situation A, to assess environmental impacts of the life cycle of a studied product. For Situation C1, existing interactions of a studied system with other systems should be accounted for using system expansion/substitution. The Situation C1 is usually related to waste management where more than one valuable co-product (e.g. electricity, heat and biogas) are co-produced. For Situation C2, it is recommended that environmental burdens of a studied system are isolated from other systems using allocation methods, e.g. physical properties (first option) and economic values (second option) of the co-products. Note that these two sub-types of decision-context situation are usually suitable for system monitoring and reporting. It is important to emphasise that time periods can be highly relevant when environmental impacts for prospective (future) product systems are modelled. A specific guideline for this aspect has not been clearly established in any LCA standards or guidelines. In the present research, prospective ALCA was used to assess environmental impacts and identify environmental hotspots for different prospective dairy scenarios in the Waikato region over one production year in 2025. First, prospective physical datasets in 2025 (e.g. prospective physical dairy farm traits, prospective transportation and prospective electricity) were predicted (extrapolated) using different techniques (see Chapter 6 for more details). Second, choices associated with co-product handling methods (e.g. allocation methods and system substitution) were tested. In general, different co-product handling methods can be optionally chosen depending on an objective of a study. For example, allocation methods can be used when environmental performance for an internal (within) dairy supply chain is targeted. However, in the case where interaction or contribution of co-products is critical or important, system substitution should be employed in order to assess the net environmental performance of a studied system. Therefore, this type of LCA modelling approach can be a useful choice to generate environmental information and support decision-making when net environmental impacts and environmental hotspots for prospective (future) product systems are studied. Specific details and applicability of this LCA modelling approach are further described in Section 7.3.3.

7.3.2 Linking decision-context situations to the present study

In this research, different research questions were formulated in order to address knowledge gaps in understanding the environmental sustainability of milk production systems in New Zealand.

In Chapter 3, multiple environmental impacts of 53 individual dairy farming systems in the Waikato region were assessed using ALCA. The environmental burdens of the cradle-to-farm gate life cycle of milk for each farm system were isolated from the rest-of-the world burdens using allocation methods. This study is equivalent to Situation C2 (Table 7.4). However, the results of contribution analysis assisted in identifying environmental hotspots to focus improvement options for dairy farming systems in the Waikato region. It is important to note that in order to verify that there are net environmental benefits from future implementation of any mitigation or improvement options, their potential consequences must be accounted for, and CLCA coupled with using system substitution to handle co-products is recommended.

In Chapter 4, the individual dairy farm systems in Chapter 3 were classified into high and low levels of farm intensification using a statistical approach. Then, a statistical comparison of multiple environmental impacts and environmental hotspots between these two farm intensification levels was carried out. As the data were simply derived from Chapter 3, again the analysis in this chapter was not directly linked to any specific decision to be supported. The results indicated that the differences in most environmental impacts (two-thirds of the assessed impact categories) between the high and low levels of farm intensification were predominantly driven by the contribution of the off-farm stages. One-third of the assessed impact categories were significantly contributed by the on-farm stage, and driven mainly by ammonia emissions (mainly from animal excreta). This implies that mitigation or improvement options for the high intensification farms (higher level of milk production per hectare) should be focussed on the off-farm stage, although maximizing resource use efficiency at the on-farm stage (e.g. feed use efficiency and fertiliser use efficiency) can obviously help reduce overall impacts.

In Chapter 5, multiple environmental impacts of different dairy farming scenarios aiming to increase milk production per hectare through the use of different farm intensification methods were assessed using CLCA. In this LCA study, future milk production was set at 20% greater than that in the baseline system (the no-action system). Three alternative intensification methods (all involving increased stocking rates) were assessed: (i) improved pasture utilization efficiency to harvest additional feed from existing pastures, (ii) increased use of N fertiliser to boost additional pasture production, and (iii) increased use of brought-in maize silage. All co-products were handled using system substitution, i.e. dairy meat and live surplus dairy calves to displace beef and surplus beef calves derived from conventional beef systems, respectively. The decision situation in this study is equivalent to Situation B, meaning that the results are intended to support decision-making. The decision question could be articulated as, "Given a demand for increased milk production, which intensification method is preferable from an environmental sustainability perspective?"

In Chapter 6, multiple environmental impacts associated with prospective dairy farming scenarios involving different farm intensification methods were compared. In this chapter, prospective dairy scenarios were set out to produce 50% more milk than the average system in the Waikato region. All prospective dairy scenarios were assessed using prospective ALCA (i.e. future scenario-based ALCA). All average flows were modified to reflect the future-average technologies. Two co-product handling methods were tested. First, environmental burdens of the cradle-to-farm gate life cycle of milk for each scenario were isolated from the rest-of-the world burdens using allocation methods. Second, system substitution to handle dairy co-products was assessed to determine its influence on the results and conclusions. The former situation is linked to Situation C2 (i.e. no decision-making to be supported), whereas the latter situation is linked to Situation A (i.e. supporting micro-level decision-making), where the associated co-product systems (e.g. conventional beef systems) were taken into account. The decision question for the former could be articulated as "Given there is a demand for increased milk production, what is the environmentally-preferred intensification option and what are environmental hotspots for the different farm intensification options?" For the latter, the decision question can be articulated as, "Given there is a demand for increased milk production, what is the environmentally-preferred intensification option and what are environmental hotspots for the different intensification options after accounting for the wider changes associated with the valuable co-products (e.g. dairy meat and surplus calves)?"

7.3.3 Use the right approaches to address the right questions

Different LCA modelling approaches can be distinguished based on their different approaches to two specific modelling aspects: (i) use of average versus marginal data, and (ii) use of allocation versus system substitution for modelling of co-products. The choices made about these two aspects will lead to generation of different sets of environmental information and subsequently result in different implications. For the first aspect, use of average data represents total environmental emissions and hence environmental impacts of a studied product system whereas use of marginal data means that only environmental emissions released from technologies (suppliers) that are actually affected by a change in demand for a studied product, or consequences resulting from an action. Therefore, results are not representative of total environmental impacts but represent only the net impacts relative to the corresponding non-action product system. For the second aspect, a separation of any interactions between a studied product system and other product systems (i.e. co-products) by using allocation methods produces potentially useful information associated with internal or within-product systems. In contrast, system substitution can generate environmental information associated with a studied product system together with the (indirect) costs and/or benefits resulting from use of its co-products. Therefore, use of these two methods generates information that must be interpreted with care for different decision situations.

In this research, the two modelling aspects were used to inform development of four types of LCA modelling approaches: (i) (current) ALCA + allocation (Chapter 3), (ii) CLCA + system substitution (Chapter 5), (iii) prospective ALCA + allocation (Chapter 6), and (iv) prospective ALCA + system substitution (Chapter 6). Table 7.5 presents comparative key characteristics between ALCA and CLCA coupled with the different co-product handling methods when applied to the life cycle of dairy farming systems.

In summary, choices associated with: (1) type of data (average or marginal), and (2) co-product handling methods, influence LCA results and therefore the implications of an LCA study. Additionally, the time frame for the decision situation should be chosen so that appropriate data can be collected for the analysis. Consequently, it is recommended that the link between the research questions and the modelling approach to be adopted is explicitly considered at the start of an LCA study. Table 7.5 provides relevant guidance for dairy farming systems based on the insights gained in this research.

This research also identified that different LCA approaches can be used in a complementary way to identify and evaluate environmental impacts (hotspots) of dairy farming systems. For example, firstly, ALCA + allocation can be used to identify environmental hotspots in dairy farming systems. This assists in identifying improvement options or mitigation options. The environmental impacts of the potential improvement/mitigation options can be evaluated (relative to each other) using CLCA + system substitution. Finally, prospective ALCA can be used to investigate total (absolute) environmental impacts of dairy farming systems after such improvement/mitigation options are implemented. Prospective ALCA can be coupled with allocation methods to handle co-products when interactions between the studied systems and companion (co-product) systems are not the focus of the study. However, prospective ALCA with system substitution to handle co-products should be used when the contribution (either benefits or disadvantages) of companion (co-products) systems are additionally of interest.

Table 7.5 Comparative characteristics and applicability of LCA approaches in pasture-based dairy farming systems in the Waikato region.

	(conventional) ALCA + allocation	CLCA + system substitution	Prospective ALCA + allocation	Prospective ALCA + system substitution
Corresponding chapter	3	5	6	6
Aim	To assess environmental impacts and identify environmental hotspots of existing (current) dairy farming systems in the Waikato region.	To assess environmental impacts associated with a change (increase) in demand for milk production through potential farm intensification methods.	To assess environmental impacts and identify environmental hotspots of different dairy systems in the future in the Waikato region. The studied system is isolated from the rest-of-the world systems (co-products) using allocation methods.	To assess environmental impacts and identify environmental hotspots of different dairy systems in the future in the Waikato region. Environmental benefits and disadvantages associated with co-products are accounted for using system substitution.
Research questions	What are the environmental impacts associated with milk from current pasture-based dairy farming systems in the Waikato region? What are the most important contributing substances for different impact categories in dairy farming systems? Where are environmental hotspots (life cycle stages) in dairy farming systems?	Which (intensification) method(s) will be the most environmentally-preferable method(s) when it has already been decided to increase total milk production (through farm intensification)? What will be the net change in environmental impacts resulting from different dairy intensification options compared with the current dairy system?	What will be the environmental impacts associated with milk from prospective (future) dairy farming systems in the Waikato region? What will the most important contributing substances be for different impact categories in prospective (future) dairy farming systems? Where will environmental hotspots (life cycle stages) be in prospective (future) dairy farming systems?	What will be the net environmental impacts associated with milk from prospective (future) dairy farming systems in the Waikato region? What will be the environmental impacts associated with the increased production of dairy co-products in the prospective (future) dairy farming systems in the Waikato region?
System boundary	A fixed cradle-to-farm gate perspective.	A cradle-to-farm gate perspective with system substitution to account for the life cycle of co-products over the life cycle of milk, e.g. dairy meat and live surplus calves.	A fixed cradle-to-farm gate perspective.	A cradle-to-farm gate perspective with system substitution to account for the life cycle of co-products over the life cycle of milk, e.g. dairy meat and live surplus calves.
Focussed functional unit	1 kg of standardized milk (e.g. fat- and protein-corrected milk), since milk is considered the function of the dairy farming systems.	1 kg of standardized milk (e.g. fat- and protein-corrected milk), since milk is considered the function of the dairy farming systems.	1 kg of standardized milk (e.g. fat- and protein-corrected milk), since milk is considered the function of the dairy farming systems.	1 kg of standardized milk (e.g. fat- and protein-corrected milk), since milk is considered the function of the dairy farming systems.
Representative flows	Average existing (current) flows	Marginal (long-term) flows	Average prospective (future) flows	Average prospective (future) flows
Co-product handling method	Partitioning or allocation, using e.g. economic values of co-products for farm inputs (inflows) and biophysical relationship for milk and dairy meat (outflows).	Avoided allocation by using system substitution. The product systems to be substituted (competing product systems) have the same functions and/or market segments to co-products to be handled, e.g. dairy meat to displace conventional beef.	Partitioning or allocation, using e.g. economic values of co-products for farm inputs (inflows) and biophysical relationship for milk and dairy meat (outflows).	Avoided allocation by using system substitution. The product systems to be substituted (competing product systems) have the same functions and/or market segments to co-products to be handled, e.g. dairy meat to displace conventional beef.

7.4 Relative versus absolute environmental indicators

In LCA, inventoried environmental emissions are characterised into different environmental impact categories based on environmental cause-effect mechanisms. One of the most useful applications of this information is that it supports users to redesign a product system in order to improve its environmental performance, i.e. reducing the magnitude of environmental impacts per unit of service delivered by a product system (Finnveden et al. 2009). However, it is clear that this environmental benefit does not always result in a net reduction in global environmental impacts since improved environmental performance of a product system may trigger increased consumption of this product, and hence increased overall impacts (Steffen et al. 2015; Font Vivanco et al. 2016). For example, improved production technologies (e.g. renewable energy-based electricity generation) can result in improved environmental performance of a product (e.g. greener electricity) together with a reduced cost price (e.g. cheaper electricity); this usually leads to increased product consumption (e.g. more electricity used) and thereby increased total environmental impacts. Therefore, this can cast doubt on the usefulness of a relative environmental sustainability indicator derived from LCA (Bjørn and Hauschild 2013; Hauschild 2015).

In 2009, a planetary boundary was proposed to define a safe operating space for human societies to properly develop and maintain the function and resilience of the Earth system (Rockstrom et al. 2009). Building on the basic framework of the safe operating space, as proposed by Rockstrom et al. (2009), Steffen et al. (2015) defined nine functions[6] (aspects) of the Earth system. Additionally, these authors quantified a safe value (control variable) for each of these aspects, and recommended an urgent need to ensure that this value is properly controlled in order for it to stay under the corresponding threshold (i.e. safe operating space).

[6] These functions refer to the biophysical processes that regulate the stability of the Earth system for humanity, comprising: (i) Climate change, (ii) Change in biosphere integrity, (iii) Stratosphere ozone depletion, (iv) Ocean acidification, (v) Biogeochemical flows (N and P cycles), (vi) Land-system change, (vii) Freshwater use, (viii) Atmospheric aerosol loading, and (ix) Introduction of novel entities (e.g. chemical pollution) (Steffen et al. 2015).

There have been attempts to introduce the planetary boundary (i.e. absolute environmental indicators) based on carrying capacity[7] (threshold for each variable control) of the Earth system for use in LCA (Sala and Goralczyk 2013; Bjørn and Hauschild 2015; Sandin et al. 2015). For example, Bjørn and Hauschild (2015) used absolute environmental indicators that were modelled based on planetary boundaries as normalisation references for use in LCA studies. This could be a promising breakthrough for future development in introducing an absolute environmental concept to develop absolute environmental characterisation models and subsequently absolute environmental impact categories at a product or sector level.

Note that all the results presented in this thesis are in a relative form i.e. the results show the comparative environmental advantages and disadvantages of different dairy farming systems (scenarios) and different farm intensification methods per kg FPCM (except for Section 7.2.2 where a per-hectare and kg-product protein basis was tested – but these are still relative values). However, this does not mean that the preferred dairy farming scenario(s) or the desired farm intensification method(s) with lower environmental impacts do lead to an absolute reduction in environmental impacts. It would be preferable to develop a method or approach that can generate absolute environmental information to support decision-making – and the recent work on planetary boundaries may provide a way forward.

In summary, relative and absolute environmental assessment methods can be used in a complementary manner. A relative environmental assessment can be useful for improving environmental performance of a studied product, while absolute environmental assessment can be used to establish global-, sector- or product-environmental limits (the two latter limits remain under development) in order to ensure that the functioning of the Earth system is properly managed and maintained. In addition, absolute environmental indicators may be used as normalisation factors to prioritise individual impact categories or as weighing factors (i.e. objective weighing factors, as opposed to the current subjective weighing factors) for calculation of a single environmental indicator for studied product systems.

[7] The carrying capacity is defined as 'the maximum sustained environmental intervention a natural system can withstand without experiencing negative changes in structure or functioning that are difficult or impossible to revert' (Bjørn and Hauschild 2015).

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Chapter 8 General Conclusions and Recommendations for Future Research

8.1 Introduction

Global consumption of milk and dairy products has increased over the past decade (FAO 2015) and is expected to substantially increase over the next decade (OECD/FAO 2015). New Zealand is the single-largest dairy exporting nation in the world and its government has articulated a vision to double its milk production and exports over the next decade (New Zealand Government 2012). However, although the dairy sector contributes positively to global food security, it can also impact negatively on the environment (FAO 2006, 2010). Therefore, it is important to optimise milk production while maintaining the overall quality of the environment. In the present research, the environmental performance of pasture-based dairy farming systems in the Waikato region (the single largest dairy region in New Zealand) was assessed using Life Cycle Assessment (LCA). Also, different potential farm intensification methods aiming at increasing milk productivity (i.e. the amount of milk produced per hectare) were evaluated using different LCA approaches.

As outlined in Chapter 1, the objectives of the present research were to:

1) Assess environmental impact profiles, and identify environmental hotspots for the cradle-to-farm gate life cycle of existing pasture-based dairy farming systems in the Waikato region of New Zealand using ALCA (addressed in Chapter 3).
2) Compare two contrasting levels of dairy farm intensification regarding their environmental impacts, and determine the relationship between key dairy farm attributes and life cycle environmental impacts of pasture-based dairy farming systems in the Waikato region (addressed in Chapter 4).
3) Explore potential farm intensification options for pasture-based dairy farming systems in the Waikato region with respect to their life cycle environmental impacts, including the interactions with other product systems such as conventional beef farming systems, using CLCA (addressed in Chapter 5).
4) Assess the environmental impacts and environmental trade-offs associated with different farming intensification options for future pasture-based dairy farming systems in the Waikato region, and investigate the influence of different dairy co-product handling methods on changes in life cycle environmental impacts of milk using prospective ALCA (addressed in Chapter 6).

In addition, the research contributed to a more general level to understanding of trends associated with dairy farming systems in New Zealand, and state-of-the-art LCA methodologies (reported in Chapter 2) and their use in supporting decision-making (reported in Chapter 7).

The key aspects (findings) of the present research are concluded in the following sections:

- The significance of the present research (Section 8.2).
- The environmental hotspots of current dairy farming systems in the Waikato region (Section 8.3).
- The environmentally-preferred intensification options for future dairy farming systems in the Waikato region (Section 8.4).
- The implications of LCA in supporting decision-making (Section 8.5).
- Recommendations for maximising performance of environmental assessment using different LCA approaches (Section 8.6).
- Recommendations for future research (Section 8.7).
- Final conclusions (Sections 8.8).

8.2 Significance of the present research

The present research is one of only a few comprehensive LCA studies on dairy farming systems where multiple environmental impacts have been assessed, and it is the only LCA study on pasture-based dairy farming systems with year-round grazing where 12 out of 15 recommended impact categories for the sustainable food production and consumption sector have been included. The exclusion of three impact categories (Resource Depletion, Water Use Impact and Land Use Impact) was due mainly to the lack of suitable data regarding inventory models/factors for these impact categories. The environmental indicators and characterization models used in this research are the most up-to-date (Hauschild et al. 2013), and are recommended by relevant international organizations, e.g. the European Commission (EC-JRC-IES 2011) and the European Food Sustainable Consumption and Production Roundtable (European Food SCP Roundtable 2013). Moreover, the study used the most recent New Zealand-specific inventory models (Ministry for Primary Industries 2013; Overseer 2014). The most recently-developed, scientific-based generic inventory models (e.g. Rotz et al. (2013);

Webb et al. (2013); Nemecek et al. (2014)) were used only when relevant New Zealand-specific models were not available.

The research generated a range of environmental information sets for pasture-based dairy farming systems in the Waikato region, derived from the use of different LCA modelling approaches. As a result, the present research has two main advantages. First, it facilitates an exploration of which LCA modelling approaches should be used and when they are most appropriate. Second, it generates different sets of environmental information to support different decision-making situations associated with intensification options for future pasture-based dairy systems in the Waikato region.

8.3 Environmental hotspots of pasture-based dairy farming systems in the Waikato region, New Zealand

Identification of environmental hotspots in the life cycle of product systems facilitates prioritisation of improvement options focused on specific processes or life cycle stages (Finnveden et al. 2009), and guides the focus of future research to develop new effective improvement options. In this research, contribution analysis was performed to identify the environmental hotspots of current pasture-based dairy systems in the Waikato region.

Table 8.1 summarises the contribution of the different life cycle stages as well as key contributing substances (environmental emissions) and their corresponding processes or activities for individual environmental indicators derived from the ALCA study presented in Chapter 3. From Table 8.1, it can be concluded that the most significant contributing substances and their corresponding processes for different environmental impact categories vary between impact categories. In addition, it illustrates the significance of some inputs and activities across a number of impact categories. In particular, manufacturing and use of fertilisers makes a significant contribution to all but one (POFP) of the impact category results. Manufacturing and/or use of fossil fuels significantly contributes to six impact category results, whereas animal excreta makes a significant contribution to six impact category results. This suggests that improvement options should focus on reducing and/or improving the efficiency of fertiliser use and fossil fuels, and improving management of animal excreta.

Table 8.1 A summary of environmental hotspots associated with key contributing substances and processes or stages in the cradle-to-farm gate life cycle of current pasture-based dairy farming systems in the Waikato region.

Impact categories	Contribution of life cycle stages (% of total impact) [a]	Contribution of environmental emissions (% of total impact) [b] and key contributing process/activity
CC	On-farm (68%) Rearing of replacement heifers (16%) Production of brought-in feeds (8%) Manufacturing of agrichemicals (6%) Transportation of off-farm inputs (2%)	CH_4 (59%) – enteric fermentation CO_2 (21%) – use of fossil fuels and N fertilisers N_2O (20%) – animal excreta (urine)
ODP	On-farm (37%) Manufacturing of agrichemicals (26%) Transportation of off-farm inputs (15%) Rearing of replacement heifers (13%) Production of brought-in feeds (9%)	Halon 1301 (66%) – manufacturing and use of fossil fuels Halon 1211 (15%) – manufacturing of fossil fuels and N fertilisers CFCs-10 (7%) – production of pesticides and fossil-sourced electricity
Non-cancer	On-farm (52%) Production of brought-in feeds (25%) Rearing of replacement heifers (15%) Manufacturing of agrichemicals (7%) Transportation of off-farm inputs (1%)	Zn (59%) – manufacturing and use of P fertilisers Cd (20%) – manufacturing and use of P fertilisers Hg (14%) – manufacturing and use of P fertilisers Pb (2%) – manufacturing and use of P fertilisers
Cancer	On-farm (46%) Manufacturing of agrichemicals (27%) Rearing of replacement heifers (10%) Production of brought-in feeds (10%) Transportation of off-farm inputs (7%)	Cr (94%) – manufacturing and use of P fertilisers Hg (3%) – manufacturing and use of P fertilisers Cd (2%) – manufacturing and use of P fertilisers
PM	On-farm (54%) Manufacturing of agrichemicals (19%) Rearing of replacement heifers (16%) Production of brought-in feeds (6%) Transportation of off-farm inputs (5%)	NH_3 (67%) – animal excreta (urine) and use of N fertilisers Fine particles (23%) – manufacturing of chemical fertilisers SO_2 (9%) – manufacturing of chemical fertilisers
IR	Manufacturing of agrichemicals (46%) Rearing of replacement heifers (21%) Transportation of off-farm inputs (21%) Production of brought-in feeds (6%) On-farm (6%)	^{222}Ra (54%) – electricity generated from nuclear power plants ^{14}C (43%) – electricity generated from nuclear power plants
POFP	On-farm (64%) Rearing of replacement heifers (14%) Transportation of off-farm inputs (11%) Manufacturing of agrichemicals (6%) Production of brought-in feeds (5%)	VOCs (56%) – animal excreta (faeces) NO_x (30%) – use of fossil fuels CH_4 (7%) – enteric fermentation
AP	On-farm (70%) Rearing of replacement heifers (18%) Production of brought-in feeds (6%) Manufacturing of agrichemicals (3%) Transportation of off-farm inputs (3%)	NH_3 (91%) – animal excreta (urine) and use of N fertilisers SO_2 (5%) – manufacturing of chemical fertilisers NO_x (4%) – use of fossil fuels
TEP	On-farm (73%) Rearing of replacement heifers (18%) Production of brought-in feeds (6%) Transportation of off-farm inputs (2%) Manufacturing of agrichemicals (1%)	NH_3 (95%) – animal excreta (urine) and use of N fertilisers NO_x (5%) – use of fossil fuels
FEP	On-farm (39%) Manufacturing of agrichemicals (25%) Rearing of replacement heifers (19%) Production of brought-in feeds (15%) Transportation of off-farm inputs (2%)	P loss (100%) – manufacturing and use of P fertilisers
MEP	On-farm (68%) Rearing of replacement heifers (14%) Production of brought-in feeds (12%) Transportation of off-farm inputs (4%) Manufacturing of agrichemicals (2%)	NO_3 (72%) – nitrate leaching originated from animal urine NH_3 (17%) – animal excreta (urine) and use of N fertilisers NO_x (12%) – use of fossil fuels
Ecotox	Manufacturing of agrichemicals (42%) Rearing of replacement heifers (21%) Production of brought-in feeds (19) On-farm (13%) Transportation of off-farm inputs (5%)	Heavy metals, e.g. Zn and Cu (70%) – manufacturing and use of P fertilisers Pesticides (30%) – use of pesticides for production of brought-in feeds

[a] Detailed definition of life cycle stages is given in Chapter 3; [b] These substances contributed at least 2% to the total impact.

Subsequently, the results derived from a statistical comparison of environmental impacts between the high and low levels of farm intensification (reported in Chapter 4) highlighted that the main hotspot driving the differences in environmental impacts between these two contrasting farm intensification levels was the off-farm stage, including the production of brought-in feed, manufacturing of agrichemicals (e.g. fertilisers and pesticides) that are used directly on a dairy farm, and transport of off-farm inputs for use on a dairy farm.

The contribution of the on-farm stage to the differences in most environmental impacts per kg FPCM between these two dairy farm intensification levels was insignificant, implying that the amounts of off-farm inputs per hectare were linearly correlated with milk production per hectare. The exception was for some impact categories (PM, POFP, AP and TEP) that were driven predominantly by NH_3 emissions (mainly from animal excreta), and for these impact categories the contribution of the on-farm stage significantly drove the differences. Note that the on-farm NH_3 emissions were mainly derived from animal urine, emphasising that interventions to reduce urinary NH_3 emissions at the on-farm stage are important, and should be prioritised. Another significant source of NH_3 emissions was the use of N fertilisers.

In contrast, when environmental indicators were characterised on a per-hectare basis, all life cycle stages for all impact categories in the high intensification group were significantly higher than those in the low intensification group. The exception was only for the impacts on Cancer, Non-cancer and Ecotox, where the contribution of the stage of rearing of replacement animals did not significantly differ between the two dairy intensification groups for the two former environmental indicators, and the contribution of the on-farm stage did not significantly differ for the latter environmental indicator. Note that these three environmental indicators were driven mainly by the contribution of heavy metals, where the characterisation models for heavy metals remained highly uncertain and therefore the results need to be interpreted with care (Rosenbaum et al. 2008; Pizzol et al. 2011). The key drivers of these differences were associated mainly with higher use of off-farm inputs per hectare (e.g. stocking rate, N fertiliser and brought-in feeds) in the high intensification group relative to the low intensification group.

In the life cycle of dairy farming systems, the off-farm activities are generally controlled by other actors in the supply chain, compared with on-farm dairy activities which are under the direct control of the dairy sector in New Zealand. This makes it more challenging for this sector to effectively address these off-farm environmental impacts. However, one of the most important interventions that is under the direct control of the dairy sector in New Zealand is to maximize resource use efficiency, i.e. reduce the use of off-farm inputs while maintaining dairy productivity (milk production).

In summary, based on the LCA results in Chapter 3, the activities that should initially be the focus of improvement or mitigation options include: (i) enteric CH_4 emissions, (ii) management of animal excreta, especially urine, (iii) use of N and P fertilisers, (iv) use of fossil energy, and (v) use of pesticides.

8.4 Environmentally-preferential intensification methods

In Chapters 5 and 6, two LCA approaches (CLCA and prospective ALCA) were used to assess multiple environmental impacts associated with three main potential intensification options involving increased stocking rates for future pasture-based dairy farming systems in the Waikato region. The intensification options were: (i) increased pasture utilisation efficiency, (ii) increased use of N fertiliser to boost on-farm pasture production, and (iii) increased use of different brought-in feeds. The conclusions with respect to an environmentally-preferred intensification option (ranking) using each of the two LCA approaches are summarised in this section.

In the CLCA study (reported in Chapter 5), the three intensification options were modelled to produce the same amount of extra milk (i.e. 20% more milk than that in the year 2010/11: 12,850 kg FPCM) and this was achieved primarily through increased stocking rates. Improved pasture utilisation efficiency was the most preferred intensification option since it resulted in relatively lower environmental impacts than the two other scenarios. As discussed in Chapter 5, the choice of the most environmentally-preferred option between the increased use of N fertiliser and the increased use of brought-in maize silage depends upon the prioritisation between, and the scale of, the different environmental impacts. Most environmental impacts driven by (or related to) N-based substances (CC, PM, AP, TEP and MEP) were higher for the increased use of N fertiliser scenario whilst environmental impacts driven by emissions during production of maize silage (ODP, Non-cancer, Cancer, IR, POFP, FEP and Ecotox) were higher for the increase use of brought-in maize silage scenario.

In the prospective ALCA study (see Chapter 6), six prospective dairy farming intensification scenarios (three of them were the same as in Chapter 5) were assumed to produce the same amount of milk per hectare and this yield was 50% higher than that in 2012. All the scenarios had increased stocking rates, although one scenario had a relatively lower stocking rate than the others since it was assumed to have relatively higher animal productivity (increased milk production per cow). The intensification methods were: (i) increased animal productivity, (ii) increased use of mixed brought-in feed, (iii) improved pasture utilisation efficiency, (iv) increased use of N fertiliser to boost pasture production, (v) increased use of brought-in maize silage, and (vi) replacement of total mixed brought-in feed in the second scenario by wheat grain. In addition, predicted progress in dairy productivity, along with involved technologies for the year 2025 (e.g. electricity grid mix), were taken into account. The improved animal productivity scenario was the most environmentally preferable, followed by the increased pasture utilisation efficiency scenario. Again, there were environmental trade-offs between the increased N fertiliser and increased use of mix brought-in feeds. The use of brought-in wheat grain was the least environmentally preferable option since it generated highest (worst) environmental impacts per kg FPCM compared with the other intensification options.

8.5 Implications of LCA in supporting decision-making

In LCA, choices associated with type of data (e.g. average versus marginal) and co-product handling methods (e.g. allocation versus system substitution) influence results and therefore the implications of an LCA study. While results derived from ALCA (based on average flows coupled with using allocation methods to handle co-products) can facilitate identification of improvement options, CLCA (based on marginal flows coupled with using system substitution to handle co-products) can assist in identifying environmental consequences resulting from a change (or an action) relative to those derived from without a change (or without an action), taking into account the contribution (benefits or disadvantages) of co-products in a wider perspective (economy). It is important to note that CLCA generates environmental information on the differences between two scenarios (i.e. with and without action). Therefore, CLCA results do not represent the total environmental impacts associated with product systems.

In the present research, prospective ALCA (based on prospective average flows coupled with using either allocation methods or system substitution to handle co-products) was used to assess the environmental impacts of future dairy farming systems. In contrast to the results derive from CLCA, this approach generates information on the total environmental impacts associated with future product systems. In order to identify environmental hotspots, ALCA coupled with allocation to account for co-products should be used, whereas in order to understand the additional environmental benefits or impacts of utilising co-products, system substitution is recommended to be used instead of allocation.

8.6 Recommendations to maximise the usefulness of LCA

This research has found that different LCA modelling approaches (which require different types of data and different co-product handling methods) can be used in a sequential and complementary way to improve decision-making. Initially, ALCA, which is based on historical average flows coupled with using allocation methods to handle co-products, can be used to assess environmental impacts and identify environmental hotspots. This information can assist in identifying improvement options. Then, the identified improvement options should be evaluated using CLCA where flows that are actually affected (marginal technologies) by such improvement options and system substitution to handle co-products should be used. Finally, prospective ALCA, based on prospective average flows coupled with using allocation methods and/or system substitution to handle co-products, should be used to assess total (net) environmental impacts derived from implementing such improvement options. Allocation methods to handle co-products can be used when the environmental contribution of the co-products is not the focus (e.g. product labelling schemes). System substitution to handle co-products can be used when the environmental contribution of the co-products (which may be either an increase or reduction in environmental impacts) is a focus and the results are expected to be interpreted in a wider perspective. It should be noted that there is no reference associated with this recommendation in the existing literature.

8.7 Recommendations for future research

In the present research, 12 out of the 15 best existing impact categories recommended by the ILCD Handbook (EC-JRC-IES 2011) and the European Food Sustainable Consumption and Production Roundtable (European Food SCP Roundtable 2013) were assessed. Therefore, it is recommended to include at least those three impact categories that were not assessed in the present research (Resource Depletion, Water Use Impact and Land Use Impact), in order to reveal any problems associated with burden-shifting between impact categories. Additionally, as models or procedures to calculate inventory data for some environmental emissions are not yet available, assumptions or simplifications were made in order to make the study as comprehensive as possible. Therefore, more detailed research on these models and methods is required. The following sections (Sections 8.8.1 to 8.8.5) recommend some significant aspects for future research.

8.7.1 More accurate inventory data

Accuracy and representativeness of data cannot only lead to improved certainty of LCA results but also to increased reliability and credibility of LCA studies (Björklund 2002; Ross et al. 2002). However, inventory models or procedures for some environmental emissions are lacking. Therefore, these emissions are usually inventoried by making simplifying assumptions.

In the present research, inventory analysis for emissions of heavy metals and pesticides was subjectively simplified. Generally, these two groups of emissions have a significant impact on human health toxicity and freshwater ecotoxicity (Rosenbaum et al. 2008; Xue et al. 2015). As a result, this simplification increased the uncertainty of the LCA results. In pasture-based dairy farming systems, these two groups of emissions contribute to environmental impacts at both the off-farm (e.g. feed crop production) and on-farm stages. The various heavy metals have different properties and therefore can reach different environmental compartments at different rates (Pizzol et al. 2011). Similarly, different active ingredients in pesticides coupled with different application technologies can lead to different quantities of emissions reaching different environmental compartments (Dijkman et al. 2012). Therefore, in order to improve the accuracy of these data, they should be inventoried using effective and reliable models which are currently under development (Rosenbaum et al. 2015).

8.7.2 Identification of actually affected product systems

To handle dairy co-products using system substitution, appropriate product systems to be substituted need to be identified. Identification of the to-be-substituted product systems is usually not straightforward (Ekvall and Weidema 2004) and is also uncertain (Weidema et al. 2009). In this research, New Zealand conventional beef products, i.e. beef and surplus live calves, were chosen as the to-be-substituted products. However, generally the market for beef is global (Dalgaard et al. 2014). Therefore, in order to improve accuracy and precision with respect to identification of the actually affected beef systems, the environmental impacts of a mix of global beef systems will have to be studied and established.

8.7.3 Consideration of additional impact categories

To date, there are a number of LCIA methods which can be used in LCA studies (Owsianiak et al. 2014; Bueno et al. 2015). In the present study, twelve impact categories and their most up-to-date characterization models were selected (EC-JRC-IES 2011; Hauschild et al. 2013). This makes the present study one of the most updated and comprehensive environmental assessment studies on pasture-based dairy farming systems. However, from the list recommended by EC-JRC-IES (2011), there are three other environmental aspects (impacts) that are not covered and addressed by the present study: (i) Resource Depletion (fossils and minerals), (ii) Land Use Impact, and (iii) Water Use Impact. Therefore, for a future environmental assessment study, at least these three impact categories should be additionally included in order to prevent potential environmental swapping.

In addition, the absolute environmental concept (i.e. planetary boundary) is a promising approach that is being developed to complement the relative assessment method used in current LCA approaches (Sala and Goralczyk 2013; Bjørn and Hauschild 2015; Sandin et al. 2015). Moreover, two other sustainability pillars (economic and societal aspects) of overall sustainability assessment (Life Cycle Sustainability Assessment) must be accounted for in order to address the overall sustainability of dairy farming systems (Kloepffer 2008; Guinée et al. 2010).

8.7.4 Consideration of spatially-differentiated characterisation models

It has been recommended that spatially-differentiated characterisation models should be integrated in LCIA methods in order to improve accuracy of LCA results, especially when assessing environmental impacts at a local scale, e.g. FEP (Morais et al., 2016; Nitschelm et al., 2016). In the present research, the FEP impact category was characterised using a generic characterisation model that was developed based mainly on European conditions where P-based substances were considered as a single contributing group (see Section 2.6.8.2 for more details) (EC-JRC-IES, 2011). However, limiting factors (e.g. N and P) for growth of phytoplankton in different freshwater ecosystems can be substantially different, subsequently causing different impacts (Elser et al. 2007). For example, in contrast to European conditions, either N-based substances or P-based substances, or both, can be limiting factors for freshwater ecosystems in New Zealand (Abell et al. 2010; Pearson et al., 2016; Smith et al., 2016). Therefore, there is a need to develop spatially-differentiated characterisation models and integrate them into LCIA methods.

8.7.5 Potential mitigation options

In the present study, the focus was evaluation of environmental impacts associated with potential choices between alternative dairy farming intensification options related to increasing pasture-based milk production in response to government policy. However, there are potential mitigation options that could be used to reduce the environmental impacts of milk (Clark et al. 2007; von Keyserlingk et al. 2013; Monaghan and de Klein 2014) but which have not been considered in the research reported here. These include: (i) increased feed crop production efficiency (van Middelaar et al. 2013; Henriksson et al. 2014), (ii) use of housing systems combined with manure management systems and relevant technologies, e.g. bioelectricity generation (Zhang et al. 2013; Battini et al. 2014), (iii) use of agricultural precision technologies, e.g. improved management of fertilisers and feed (Sova et al. 2014; Borchers and Bewley 2015), and (iv) improved animal health and longevity (Williams et al. 2015; Chen et al. 2016). However, these options must be evaluated taking into account their consequences (both upstream and downstream) along the life cycle of milk to ensure net environmental improvement.

8.8 Final conclusions

Life Cycle Assessment is a useful system-level analytical approach for assessing the comprehensive environmental impacts of dairy farming systems. However, it is important to recognise that different LCA modelling approaches generate different sets of environmental information, leading to differences in interpretation of results and thereby implications for decision-making. Therefore, the choice of a particular LCA modelling approach needs to be tailored to the question of concern and the intended use of the results.

The present research has demonstrated that pasture-based dairy farming systems in the Waikato region cause a range of environmental impacts. In the absence of effective interventions (e.g. farm management practices, strategies or policies), and based on the current trends in New Zealand dairy systems, many environmental impacts in 2025 are likely to be substantially worse when compared with the current dairy systems (on both a per kg fat- and protein-corrected milk and per hectare basis). However, different dairy farm intensification options result in different environmental benefits and disadvantages. This research has highlighted that improved animal productivity (increased milk production per cow while maintaining the same amount of inputs) and increased pasture utilisation efficiency are two of the most promising intensification options for pasture-based dairy systems in the Waikato region. Apart from these two options, this research indicated that other intensification methods based on increased use of on-farm N fertilisers and use of brought-in feeds all lead to an increase in most categories of environmental impacts, compared with the current dairy systems.

However, the selection of alternative dairy intensification options need to be undertaken as part of a decision-making process that involves evaluation of the relative significance of different environmental impacts and their contributing activities. This significance could be based on the contributions of different environmental indicators to exceedance of the corresponding indicators at the level of planetary boundaries.

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