ANALYSIS OF RESOURCE USE AND GREENHOUSE GAS EMISSIONS FROM FOUR AUSTRALIAN MEAT PRODUCTION SYSTEMS, WITH INVESTIGATION OF MITIGATION OPPORTUNITIES AND TRADE-OFFS

by

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PERSONAL STATEMENT

This thesis integrates the things I love about life – food, agriculture, livestock and the pursuit of knowledge, together with a genuine desire to improve the world for our children. Coming from a third-generation Australian farming family, I regard the production of food as a noble task, as is the vital work of improving food systems and minimising the negative impact they have on the environment. The world is not here for us to exploit, but to care for, and when done well, this will be for the betterment of agricultural systems, natural ecosystems, and humanity. I have always felt that we can, and must, balance commercial gain with the needs of the animals and land we manage and the broader environment, and I have carried these ideals into my research. I believe that the intricate and complex world in which we find ourselves points to a creator God, to whom I owe gratitude for the privilege of stewardship. This thesis seeks to distil the results of 10 years work, and I hope that it will provide a building block for better understanding and management of Australia’s agricultural systems into the future.
ABSTRACT

One major challenge of the 21st century is to increase food production to meet the needs of a growing global population, while minimising negative impacts to the environment. This study conducted a series of attributional life cycle assessment (aLCA) studies, to determine greenhouse gas (GHG) emissions, energy demand, water use, water stress, crop land occupation and human edible protein required (HEPR) for Australian meat products. The study applied consistent methods and system boundaries to establish benchmarks, determine impact hotspots, trade-offs and mitigation opportunities. The effect of different production systems and regions, and the sensitivity of GHG emission methods was also investigated. Knowledge gaps and future research needs have been identified. The results provide the first consistent, multi-industry dataset covering Australian beef, lamb, pork and chicken meat, enabling analysis of issues that are relevant to multiple industries. Foreground data from more than 80 case study farms (CSF) and 19 meat processing plants were compiled and augmented with regional (RAF) or national scale farm datasets. Results were standardised to report impacts relative to one kilogram of boneless, fat corrected meat. The results showed that GHG emissions were high for beef, lamb and wool in comparison to pork and chicken meat. In contrast, crop land occupation and human edible protein required (HEPR) was high for pork and chicken meat, revealing a significant trade-off between these indicators. Smaller differences were observed between species with respect to energy demand, and water impacts were found to be highly regionally sensitive. Across all meat production systems, water impacts were lower than reported previously in global analyses. Mitigations were identified that reduced environmental impact across multiple impact categories, while others showed that substantial trade-offs between different impact categories. Further research and extension should focus on strategies that reduce multiple impacts rather than focusing on only one area, such as GHG emissions.

This thesis demonstrated the substantial contribution made by this body of work to GHG emission estimation methods in the Australian National Inventory Report (NIR) and to global livestock LCA methods. This research focused on meat production systems through to production of a wholesale product. Further research is required to cover the full ‘cradle-to-grave’ of meat production, consumption and...
disposal, to inform consumers of the impact of meat consumption. Where information is sought to inform a choice between one meat and another, new analyses using cLCA methods are required. To advance this, there is an urgent need for new methods, data and research capacity to be developed in this area.
DECLARATION

I, Stephen Geoffrey Wiedemann, certify that this dissertation or thesis does not incorporate, without acknowledgement, any material previously submitted for a degree or diploma in any university. It does not contain any material previously published or written by another person except where due reference is made in the text.

Co-author declaration statements are provided to clearly explain the contributions by the author and co-authors, as well as permission from co-authors to publish these papers for the purpose of this thesis (see Appendix 1).

Signed:

[Signature]

Stephen Geoffrey Wiedemann
3rd November, 2018
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There are many who contributed to the research contained in this thesis over the past decade, and invariably I will not acknowledge all who have assisted here. However, I will endeavour to recognise the many people who contributed to this research and to my professional development in general.

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To my young children, who will remember the time when I worked on this thesis as a period of significantly less time with their dad, I hope that when you are older you will appreciate that the pursuit of knowledge, truth and working towards better food production systems is worth dedicating significant effort to, even when it requires sacrifice of things we love!

Lastly, I can’t sufficiently state my gratitude to my wife, Madeleine Wiedemann, who is a partner in all aspects of my life. Your support over some painful years of work, your positivity, and your practical assistance in proofing this document has been incredible. One day I’ll repay the effort when you embark on a Ph.D!
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GLOSSARY OF ABBREVIATIONS

ABARES Australian Bureau of Agricultural and Resource Economics and Sciences
ABS: Australian Bureau of Statistics
AD: anaerobic digestion
ADG: average daily gain
AECL: Australian Egg Corporation Limited
AWI: Australian Wool Innovation
CFI: Carbon Farming Initiative
CP: crude protein
CRC: Co-operative Research Centre
CSF: case study farms
CSIRO: Commonwealth Scientific and Industrial Research Organisation
CW: carcase weight
d: days
DAFF: Department of Agriculture, Fisheries and Forestry
DCCEE: Department of Climate Change and Energy Efficiency
DEE: Department of the Environment and Energy
DEFRA: Department for Environment, Food and Rural Affairs
dLUC: direct land use change
DofE: Department of the Environment
DSEWPaC: Department of Sustainability, Environment, Water, Population and Communities
ERF: Emission Reduction Fund
EU JRC: United Nations Joint Research Commission
FAO: Food and Agriculture Organisation (of the United Nations)
FCR: feed conversion ratio
FR: free range
FU: functional unit
GEI: gross energy intake
GHG: greenhouse gas
HEPR: human edible protein requirement
HEPR-CE: human edible protein requirement conversion equivalent
HSCW: hot standard carcase weight
ILCD: International Reference Life Cycle Data System
IPCC: Intergovernmental Panel on Climate Change
ISO: International Organisation for Standardization
INTRODUCTION

1.1 ENVIRONMENTAL SUSTAINABILITY AND FOOD PRODUCTION IN AUSTRALIAN AGRICULTURE

The ongoing ability of the world to produce and distribute food to the population in a way that minimises damage to the environment is increasingly being challenged from many different directions. Global population projections of 9.1 billion people by 2050 correspond to a 34% increase in annual demand for cereal grain (from 2.1 to 3 billion tonnes worldwide) according to the FAO (Food and Agriculture Organisation of the United Nations, 2009). In the same timeframe, the FAO (2009) predicts global meat demand to increase 74%, to 470 million tonnes. Historically, increased food production has been accompanied by expanding the area of agricultural land use (Tilman et al., 2001). However, according to the FAO (2009), available additional arable land to meet future food demand is constrained to an estimated 5% expansion, and both water and energy are increasingly constrained. Considering this, optimising resource utilisation and intensification of agricultural production is an important means of increasing food production. Moreover, in Australia, climate change is projected to reduce rainfall (Delworth and Zeng, 2014; Freund et al., 2017; Murphy and Timbal, 2008), cause higher temperatures and increased frequency of extreme high temperatures across major agricultural regions (Garnaut, 2008; Murphy and Timbal, 2008) and greater frost incidence in some regions (Zheng et al., 2015). These changes will all negatively impact food production, particularly in southern Australia. These changes may be offset to some extent by positive impacts arising from carbon dioxide fertilisation (Cline, 2007; Kimball, 1983) and reduced or delayed incidence of frost in some regions (Zheng et al., 2015). Nonetheless, researchers agree that reducing greenhouse gas (GHG) emissions as a means of mitigating climate change is a priority (Johnson et al., 2007; Smith et al., 2008) in addition to adaptation (Anwar et al., 2007; Howden et al., 2007).

Clearly, the world’s food producers face a significant challenge to ensure that sufficient food is produced to feed the world's population. The FAO (1996) defined food security as the state in which ‘all people, at all times, have the physical and economic access to sufficient, safe and nutritious food to meet their dietary needs.
and food preferences for an active and healthy life.’ This definition indicates the complexity of the issue, combining economic and social factors. Indeed, food security is influenced by problems with food distribution to a substantial extent and this is an important, global problem (Pinstrup-Andersen, 2009). However, environmental sustainability is not concerned with producing food for human consumption only, but also takes into account the needs of the environment. Environmental sustainability has been described in various ways including ‘ecological stability, economic viability and socio-cultural permanence’ (Lal, 1991), or more broadly as the ‘maintenance of natural capital’ (Goodland, 1995). Environmental sustainability must also incorporate the protection of unique plant and animal species in their own rights, separate from their perceived value to humans (Tilman et al., 2001). In Australia, Ecologically Sustainable Development principles have been developed and applied to agricultural sustainability (ESDSC, 1992). Ecologically Sustainable Development is defined as ‘using, conserving and enhancing the community's resources so that ecological processes, on which life depends, are maintained, and the total quality of life, now and in the future, can be increased’ (ESDSC, 1992). These principles provide broad guidance for understanding sustainability in Australian agriculture.

The food system is complex, involving production, distribution, processing, retailing, consumption and disposal, often in transnational supply chains. While the impacts of food production and particularly livestock can be high, other non-essential impacts also arise from activities such as air transport and vehicle use (Wynes and Nicholas, 2017) and at the high level this must be stated for context. Food is essential, and impacts are inevitable, but it is imperative that these are reduced by better management practices.

In food systems, primary production, more than any other stage in the food supply chain, operates at the interface with agricultural ecosystems, and consequently generates a high proportion of the total impacts. At the time that the research presented here was initiated (in 2008, see chapter 2.5), there was a paucity of Australian research that had investigated multiple industries or food types across the whole supply chain with consistent research methods to determine the basic resource demands and impacts generated by food products.
In the global context, Australia has historically produced small volumes of grain (< 2% of global cereal grain production) and livestock products (< 1.5% global meat production) (FAS/USDA, 2017a; FAS/USDA, 2017b); however, it has been an important contributor to the global grain and meat trade. Australia is the world’s third largest beef exporter, contributing some 15% of globally traded beef in 2016 (FAS/USDA, 2017a). Grain exports also contributed in the order of 10% of globally traded wheat (FAS/USDA, 2017b). Unlike many large exporting nations or regions such as the USA and the EU, Australia exports a large proportion of its total annual production. Both wheat and beef exports represented over 50% of Australia’s annual production, making it a unique contributor to global food supply. Australia’s ability to maintain this ongoing contribution to global food production requires maintenance of natural capital and consideration of ecosystem health. Primary issues of concern include the contribution of animal agriculture to climate change via GHG emissions (Steinfeld et al., 2006), primary energy use, land occupation and transformation. Livestock have been implicated because of their apparent contribution to very high water use (Hoekstra and Chapagain, 2007; Pimentel et al., 1997) but the impacts of using water, and particularly green water, have not been examined in detail for Australia’s livestock industries.

Life cycle assessment (LCA) is one research tool that can assess multiple impacts across supply chains, and therefore can provide unique insights into the environmental sustainability of food production (Roy et al., 2009). Importantly, LCA reports impacts relative to production (an intensity metric), which enables benchmarking of resource use and environmental efficiency, and can demonstrate where improved productivity, such as improved feed conversion, may lower environmental impacts (de Boer et al., 2011). LCA is a well-developed research tool, which is conducted in four stages, as described in the international standards (ISO - International Organisation for Standardization) (ISO 2006:14040 - ISO, 2006a; ISO, 2006b):

1. Goal and scope establishment,
2. data collection (life cycle inventory),
3. life cycle impact assessment (LCIA), and
4. interpretation.
The general LCA research method has been described in detail elsewhere (i.e. EU JRC, 2010; Guinée, 2002). However, agricultural LCAs have unique aspects: many of the impacts studied cannot be directly measured, the production cycle can be quite long (several years in the case of beef) and, particularly in Australia, production can be highly variable in response to climate. Thus, an agricultural LCA requires an in-depth understanding of agricultural production systems. An outline of the development of agricultural LCA methods in Australia is provided in the following section.

1.2 DEVELOPMENT OF LIVESTOCK LCA RESEARCH AND METHODS IN AUSTRALIA

Australian LCA research in the livestock industries was initiated in the early 2000s. In the first comprehensive review of Australian LCA studies and methods (Harris and Narayanaswamy, 2009b), only three major Australian LCAs were cited; dairy (Lundie et al., 2003), grains (Narayanaswamy et al., 2004), and sugar (Renouf, 2006). The first LCA study of red meat was commissioned in 2005 and completed in 2009 and was therefore contemporary to the review of methods (Harris and Narayanaswamy, 2009b). This work was done by a consortium of researchers, including the author of this thesis (Peters et al., 2010a; Peters et al., 2011; Peters et al., 2010b).

Despite the existence of ISO standards for LCA, the review of Harris and Narayanaswamy (2009b) identified a lack of consistency in agricultural LCA research as a limitation for Australian research. Specifically, issues relating to the choice of system boundaries, allocation, impact categories covered and data completeness made comparison difficult or impossible. To address this concern LCA guidelines were developed with support from a consortium of Australian agricultural Research and Development Corporations, to harmonise the methods and scope of agricultural LCA in Australia. The guidelines were published in 2009 (Harris and Narayanaswamy, 2009a) and preceded a series of LCA studies that contribute to this thesis, along with contemporary studies by other researchers. Key scope aspects
specified in those guidelines which shaped the majority of agricultural LCA research in Australia in the following years include:

i. Selection of an attributional LCA approach.

ii. Selection of water, energy and GHG emissions as primary impact categories to be covered.

iii. Inclusion of GHG emissions and/or carbon sequestration from land use (LU) and land use change (LUC, later termed ‘direct land use change’, or dLUC).

iv. Inclusion of water use, noting the need for new methods in this area.

v. System boundaries that covered the whole supply chain, including coverage of multiple years of data where required, to limit variation in response to climate.

vi. Inclusion of uncertainty analysis to improve the stated accuracy of the results.

The method did not provide normalisation factors for Australian conditions. While these have been developed elsewhere, normalisation decisions are preference based to some degree. While normalisation simplifies communication, sound mid-point data are required in the first instance. Thus, as a starting point, the analysis focused on mid-point assessment only. The development and completion of this methodology stimulated livestock LCA research, and largely shaped the research in this thesis, as described in the following section.
2 THESIS OVERVIEW AND SUMMARY OF RESEARCH METHODS

2.1 BACKGROUND AND RESEARCH CONTEXT

Completion of an Australian methodology for agricultural LCA (Harris and Narayanaswamy, 2009a), discussed in the previous section, enabled LCA studies of pork, chicken meat, egg, beef, lamb, wool and dairy products to be commissioned with harmonised goals and methods. The publications contained in this thesis were part of this co-ordinated research, and represent the culmination of the author’s contribution to this work, 10 years after the first study was commissioned. This research was not commissioned with the aim of demonstrating which industry was ‘best’ or even better, but it was intended to provide comparability to improve the broader knowledge base regarding livestock production in Australia, while delivering industry-specific benefits.

Because this thesis is a Ph.D by prior publication, some explanation of the context, the initial studies and contemporary research is required to understand the nature of the work. The thesis outlines the goals identified in the commissioned projects and outlines how these goals were achieved over time. This thesis is atypical compared to usual biological sciences research, because it was completed over 10 years and covered a broad area of work. Four major industries (chicken meat, pork, lamb and beef), and four major impact areas (GHG emissions, energy, water and land) were studied simultaneously, although the research could have easily been narrowed to one or two of these areas. This would have had the advantage of allowing some aspects of the research to be done in much greater depth but would have required other areas to be omitted. Single impact, single species livestock LCA studies have dominated LCA research since 2010, increasing at a greater rate than multi-impact studies, multi-species studies (McClelland et al., 2018). However, a strength of LCA is integrated research, focusing on the whole system, and consequently the focus was maintained on all industries and four impact categories. This was necessary to address the goals identified by the industries and the scope outlined by Harris and Narayanaswamy (2009b) which, identified three impact categories at a minimum.
The research was completed in two distinct stages, consisting of i) initial studies, meaning the first studies commissioned to the author in each industry and included in the supplementary materials, and ii) published studies, meaning the publications, which utilise much of the material from the initial studies, but also extend the scope and update the methods in some cases. Two important objectives (identification of sensitive and uncertain assumptions and research gaps) were addressed by the initial studies and were not featured to the same extent in the published studies. While the detail of these studies doesn’t appear in the body of the thesis (they are provided in the supplementary material), the key findings are summarised in section 2.5, together with a research timeline. This was necessary to explain the contribution of these studies to the relevant objectives and demonstrate how the research program that spanned 10 years guided the development of a much broader GHG emission research portfolio that was ultimately integrated into the published studies which form the basis of this thesis.

2.2 GOAL AND SCOPE OF THIS THESIS

The published studies were commissioned with the goal of determining environmental impacts and resource use associated with Australian meat products, including all impacts across the supply chain, using consistent methods and system boundaries to enable comparability. The target audience included government, industries and the general public. Based on this work, the aims of the thesis are:

i) Establish consistent, supply chain environmental benchmark data covering GHG emissions, energy and water use in meat production systems used in Australia, using common methods, assumptions, datasets and system boundaries. This was expanded by the author to include water stress, land occupation and human edible protein requirement (HEPR) in the published studies or in the discussion section of this thesis.

ii) Determine impact hotspots, mitigation opportunities and trade-offs associated with the major impacts from the major livestock production systems, including primary processing.

iii) Examine the effects of different production systems and regions on environmental impacts and resource use.
iv) Examine the sensitivity of key assumptions relating to greenhouse gas emission prediction and provide recommendations and guide research to quantify and reduce these impacts.

v) Critically analyse the limitations, knowledge gaps and future research requirements associated with LCA research and GHG emission accounting methods for Australian livestock systems.

The studies used an attributional LCA (aLCA) approach, which aligned with the goal of benchmarking and hotspot analysis and with global guidance for livestock LCA (LEAP, 2014; LEAP, 2015a; LEAP, 2015b; LEAP, 2016). Limitations associated with this methodological choice are discussed in section 10.7.

2.3 THESIS STRUCTURE

The thesis consists of a review of the contemporary Australian research (section 2.4), a review of the initial LCA studies (section 2.5) that preceded the published LCA studies and provides relevant context, followed by eight published LCA papers that form the body of the thesis, covering beef, lamb, wool, chicken meat and pork. Author attribution statements associated with each chapter are included in Appendix 1. These chapters are followed by a discussion and a conclusion chapter. Additional detail has been provided in Appendix 2 where this was too lengthy for the discussion chapter, and further to this, an electronic supplementary file has been included to provide copies of other relevant papers and reports by the author that provide important context.

2.4 CONTEMPORARY AUSTRALIAN STUDIES

This section contains a critical review of the contemporary, published Australian research to identify important contributions from the contemporary, published Australian LCA literature that were incorporated into the author’s research, together with limitations and knowledge gaps relevant to this thesis. The results also provide important context for the published papers in chapters 3 to 9. Note that as the scope...
of the research in this thesis was Australian livestock, the review focused on Australian studies only. International studies are considered in each published study in comparison to the Australian results.

2.4.1 BEEF STUDIES

A review of the published literature revealed three LCA studies of beef production, published between 2010 and 2012, under Australian conditions. Peters et al (2010a; 2011; 2010b) in which the author participated, considered impacts associated with GHG emissions, energy, water use and soil indicators, Ridoutt et al. (2012a; 2011) studied impacts from GHG emissions, water use and water stress, while Eady et al. (2011) studied impacts from GHG emissions and water use. Wiedemann & McGahan (2010) also conducted a review of methods used for determining supply chain water use in LCA, providing important background context.

Peters et al (2010a; 2011; 2010b) completed a LCA of two case study beef farms located in New South Wales (grain and grass finished beef) and Victoria (organic beef). The study included a broad range of LCA indicators, including GHG emissions, energy and water use, and also applied novel methods for assessing soil acidification and nutrient management. As the most cited Australian study, detailed review was warranted. The study included two years, ostensibly a ‘normal’ and ‘drought’ season. Livestock production data and farm purchases were collected from farm records. Emissions were modelled using NIR methods (Commonwealth of Australia, 2008), but excluded LU and dLUC. The study was an early livestock LCA and had novel and useful aspects. For example, a hybrid ‘process data’ and ‘input-output’ dataset was applied to avoid so-called inventory ‘truncation’ (Crawford et al., 2018), and this was found to have a minor effect (<9.2% increase in impacts) for beef and lamb.

However, limitations existed in this study because of the system boundaries applied. Herd numbers were not stable because of the short data collection period, and breeding impacts associated with purchased cattle were excluded because of the choice of a ‘farm gate’ system boundary. As a result, not all livestock emissions were
included, and some emissions intensity values were unusually low. For example, the Victorian farm had an emission intensity of 11.6 kg CO₂-e kg HSCW⁻¹ (hot standard carcase weight) in 2002 and 18.1 kg CO₂-e kg HSCW⁻¹ in 2004, but this difference corresponded to a change from purchasing young cattle and finishing (i.e. a gate-to-gate system boundary) to a breeding system (cradle-to-gate). Thus, absolute values from only one farm in one year (VIC 2004) have a complete herd inventory. Energy for the VIC 2004 year was 20.2MJ kg HSCW⁻¹. The study compared grain and grass finishing in the NSW supply chain, and showed a 17.5% lower carbon footprint for grain finishing. This supply chain also included some purchased cattle at the farm stage, which will have had the effect of reducing the absolute impacts. However, relative differences between grass and grain finishing are less likely to be influenced by this. Ridoutt et al. (2011) completed a study for six regional beef production systems in NSW, using ‘theoretical’ farm models constructed from livestock production and input data from gross-margins to estimate GHG emissions and did not utilise complete or representative farm inventories. GHG emissions ranged from 10.2 – 12.7 kg CO₂-e kg LW⁻¹ (liveweight) but differences in the herd assumptions confounded the comparison of market types and finishing types (grain and grass). Consequently, this study also provided broadly indicative results only. Impacts from LU and dLUC were not included on the premise that impacts had not occurred in the regions studied. However, the third scenario of Ridoutt et al. (2011) included production from western NSW (Walgett region), where land clearing is known to have occurred in the 20 years preceding the study (Dean et al., 2015), and consequently impacts from dLUC are expected to be important for this region.

The third study to assess GHG emissions and water was Eady et al. (2011), who completed a study of beef production on two Queensland (QLD) farms. The study was a detailed farm scale analysis of the livestock system and GHG emissions. The two case studies differed in region and market focus, with one being focused on heavy grass-fed steers (600kg LW) and one focused on trade cattle (weaner steers, 239kg LW). The study showed that weaner production had much higher emissions intensity (17.5 - 22.9kg CO₂-e kg LW⁻¹) than production of heavy finished cattle (11.6 - 15.5kg CO₂-e kg LW⁻¹), which related to the light turn-off weight of the weaner cattle, and highlighted the need to compare results at the same point in the production system. Results for the heavy cattle were 14-41% higher than results from
Ridoutt et al. (2011) and the VIC case study of Peters et al. (2010a) when results were standardised to a LW basis. These differences partly relate to differences in the herds studied, and partly to the enteric methane prediction model applied by Eady et al. (2011) for northern cattle, which generated significantly higher impacts than the prediction method for southern cattle. Subsequent research has found that enteric emissions from both regions can be predicted using the same method (Charmley et al., 2016) indicating that this was a methodological factor not an actual difference. Eady et al. (2011) also applied a novel allocation method, allocating impacts between different sale categories of livestock based on sale price (economic allocation). This approach caused different results compared to long form market data because of market prices. Market prices are also unstable over time, for example, sale prices (per kg of LW) for cull cows were 9% higher than for weaner steers at the Gympie farm. In comparison, sale prices from Queensland’s largest market (Roma saleyards) revealed a discount of 42% for cull cows compared to weaner steers in the 3 years from 2015-2017 (MLA, 2018). This shows results based on economic allocation may change over time causing variability in benchmarking results. The decision to allocate on relative price also raises philosophical questions: should environmental impacts be linked to price (driven by product quality) or simply relate to the mass of the product (which is causally linked to the provision of nutrition for human food). Eady et al. (2011) made an important contribution to the research by including estimates of tree planting required to offset livestock emissions, but historic LUC emissions from tree clearing were not discussed explicitly. With these differences noted between the studies, the results provided two case study farms (CSF) where sufficient detail exists to verify the results, but method differences make comparison difficult.

All studies also included an analysis of water use, but with different methods. Peters et al. (2010b) completed a farm-scale water balance and described both total water flows and ‘extracted’ water flows using definitions that aligned with the ABS water accounts (ABS, 2005), with impacts ranging from 27 - 540L kg HSCW⁻¹. This method did not include rainfall captured in farm dams in the definition of ‘extracted’ water and consequently reported low levels of water use for non-irrigated beef, compared to the contemporary studies (Eady et al., 2011; Ridoutt et al., 2011). Eady et al. (2011) included an assessment of water use using a water footprint method. The
blue water use (broadly equivalent to fresh water consumption) ranged from 51 - 96L kg LW⁻¹, but no detail was provided to describe the treatment of evaporation from farm dams. The study of water use by Ridoutt et al. (2012a) advanced the analysis of on-farm water use by including an operational method to estimate cattle drinking water use, and by including water use and losses from farm dams. Water use ranged from 24.7 to 234 L kg LW⁻¹, with the largest values being in response to the inclusion of irrigation water use for pasture production. These water use values all applied different methods with different inclusions and exclusions, making comparison impossible. As a consequence, these studies provided some methodological advances, but could not be considered benchmarks of water use.

Conceptually the methods developed by Ridoutt et al. (2012a) were important advances, though the study applied a theoretical approach to determine the characteristics of the farm water system, including the proportion of water sourced from dams, and the number, size and surface area of the dams, all factors that have a substantial impact on the water supply balance. Ridoutt et al. (2012a) also treated excreted water (urine and manure) as a ‘return’ and therefore did not consider this a consumptive use. However, treating this water as a ‘return’ does not satisfactorily reflect water dynamics in a farm situation. Water entrained in rainfall runoff and captured by a farm dam occurs during major rainfall events, while excreted urine and manure is deposited in small, dispersed volumes daily. These are unlikely to contribute significantly to runoff or drainage because of the typically high levels of evapo-transpiration relative to rainfall in Australia. Water excreted in manure and urine would be more accurately considered a flow to pasture, analogous to irrigation, and therefore a consumptive use. Ridoutt et al. (2012a) also applied the impact assessment method of Pfister et al. (2009) providing the first such results for Australian livestock systems.

Peters et al. (2010a) was the only early Australian red meat study to include the impacts of meat processing. Inputs and outputs associated with the full beef processing operation (i.e. slaughter, slicing, boning and chilling losses) were included. The study was reported on a HSCW basis to align with industry terminology regarding the sale of cattle to meat processors. However, the inventory aligned more correctly to a reference flow ‘out’ of the processing plant, i.e. chilled,
boxed beef. This slight misalignment of reference flows and system boundaries is common in LCA (see de Vries et al., 2015), where ‘farm gate’ results are reported on a carcase weight (CW) basis, with no assessment of transport, processing impacts or allocation to important co-products such as leather, edible offal and rendering products. The study applied a mass allocation approach to handle co-products from meat processing, which had the effect of allocating a high proportion of impacts to co-products and waste products, substantially reducing the apparent impacts per kg of beef.

2.4.2 LAMB AND WOOL STUDIES

Five studies have been completed investigating lamb and/or wool production under Australian conditions, with three focused solely on GHG emissions (Biswas et al., 2010; Brock et al., 2013; Eady et al., 2012), one on GHG emissions, energy and water use (Peters et al., 2010a; Peters et al., 2010b) and one solely on water use (Ridoutt et al., 2012b). One additional multi-impact study of wool and lamb production was completed by the author and colleagues to investigate the impact of different methods for handling co-production (Wiedemann et al., 2015b) but results are not discussed as the same data were reported in more detail in the published study in chapter 7.

The first published studies in Australia investigated impacts from lamb production from a case study in Western Australia (WA) (Peters et al., 2010a; Peters et al., 2010b) and lamb and wool from a Victorian case study (Biswas et al., 2010). Both studies investigated Merino and Merino-cross breeds. Methods applied by Peters et al. (2010a; 2010b) that were specific to sheep include the selection of economic allocation between wool and liveweight, which resulted in relatively high proportions of the environmental burden being allocated to wool and lower proportions to lamb. Impacts were 10.2 - 10.8 kg CO2-e, primary energy of 42.2 - 47.9MJ, and water use of 136 – 214L kg HSCW⁻¹.

The study by Biswas et al. (2010) studied wool production in a research trial rather than a commercial farm, and excluded emissions associated with breeding from the
system boundary. This study applied economic allocation to divide between wool and sheep meat, and the reported impacts were 4.6kg CO$_2$-e kg sheep meat$^{-1}$ (LW) and 13.9kg CO$_2$-e kg wool$^{-1}$.

Eady et al. (2012) completed a study of a mixed sheep-wheat farm in WA, presenting results for wool and grain, together with stud sheep and sheep sold for slaughter. The study presented a novel biophysical method for allocating impacts between multiple systems operating on the farm, and also applied economic allocation methods. This study contributed meaningfully to the important discussion around methods for allocating impacts between wool and sheep meat. Eady et al. (2012) proposed a biophysical method which treated wool as the primary product from the system, allocating the impacts associated with maintenance of the breeding flock to the wool product and attributed direct additional requirements associated with meat production to LW. Because the GHG emissions burden associated with maintenance of the sheep flock is considerable, this resulted in high impacts for wool and lower ones for sheep meat. The results were quite sensitive to this assumption (as later shown in Wiedemann et al., 2015b) and the choice regarding how a biophysical method separates emissions includes a degree of preference from the researcher. As with Eady et al. (2011) an economic allocation method was also applied, which separated the output of the sheep flock into different market types and allocated impacts on an economic basis. This resulted in lower impacts for ‘cull’ animals (approx. 2.7kg CO$_2$-e kg LW$^{-1}$, assuming ewes were 55kg LW at sale) and higher impacts for young animals (6.2kg CO$_2$-e kg LW$^{-1}$, assuming sale weights of 40kg LW). The biophysical allocation method resulted in lower emissions of between 2.6-3.7kg CO$_2$-e kg LW$^{-1}$ and corresponded to very high allocations to wool. The results using economic allocation were within the range of those reported by Peters et al. (2010a) depending for lambs, but the values for cull ewes was much lower. This was partly because of the differentiation between sheep types in the allocation process (not done by Peters) and possibly also because Peters et al. (2010a) studied systems with fast-growing, cross-bred lambs.

Brock et al. (2013) studied wool and lamb production from sheep production on a research station in Yass, southern NSW. The study used actual data from the research station, combined with extensive information from records kept at the site. Predicted
GHG emissions were found to be 24.9kg CO₂-e per kg of greasy wool and 6.2kg CO₂-e kg LW⁻¹ for lambs. The results for lamb were similar to the economic allocation results of Eady et al. (2012) and Peters et al. (2010a). This study highlighted that different feed intake models and enteric methane prediction models produced results that varied by 27%. Specifically, the biophysical model “GrassGro” (Moore et al., 1997) predicted higher feed intake compared to the NIR (Commonwealth of Australia, 2015) method resulting in higher predicted enteric methane emissions. This study also showed that changing enterprise mix to increase the proportion of ewes joined to terminal sires could reduce relative impacts for wool, when an economic allocation approach was applied.

Ridoutt et al. (2012b) studied water use and water stress associated with lamb production from three Victorian farms and a meat processing plant, and extended this supply chain through to the USA. This study contributed some further knowledge regarding water use from lamb production and reported impact assessment results for water stress. The study determined that drinking water use was the largest single impact throughout the supply chain, though it was noted that irrigation was not included. Considering the importance of drinking water, it is important to note that the authors did not report inventory data regarding the volume or surface area of farm dams relative to livestock numbers, and simply estimated evaporation from farm dams as a multiplier (4/3) of estimated livestock drinking water requirements. The study demonstrated that drinking water requirements were a source of considerable uncertainty in the study. The study used economic allocation to separate impacts between wool and lamb.

2.4.3 OTHER LIVESTOCK STUDIES

Only one peer reviewed study of poultry has been completed in Australia and no pig studies have been published in Australia. Bengtsson and Seddon (2013) published a study of chicken meat produced by the Ingham’s chicken company Australia wide. This was a broad study in terms of geographical coverage and impacts assessed, and was based on robust industry data for bird performance, feed types used and meat processing data. The study applied economic allocation to handle co-products at the
point of slaughter. The accuracy of the estimated impacts from this study was limited by several assumptions applied by the authors. Primary data specific to the grow-out stage was not applied to determine energy use, which was subsequently found to contribute substantially to total energy and GHG emissions. Manure ammonia emission estimates were not specific to bird diets or performance and NIR methods (Commonwealth of Australia, 2013) were applied to predict GHG emissions without sensitivity or uncertainty analysis.

The study used generic grain inventory data and it was unclear how feed additives were modelled. The study implicated ammonia as a significant environmental issue from Australian chicken meat production because of apparent contributions to eutrophication and [atmospheric] acidification. However, without sufficient detail regarding how these emissions were predicted it was not possible to compare with contemporary factors recommended for prediction of ammonia in Australia (i.e. NPI reporting, FSA, 2007). It is also notable that, while ammonia is a reportable emission in Australia (DSEWPaC, 2013), the incidence of environmental harm arising from ammonia emissions in Australia is low (Murray, 1989) compared to Europe, where acid rain is a well-documented environmental problem arising from ammonia emissions (Kruse and Bell, 1987; Menz and Seip, 2004). The study included land use and land transformation, but methods, inventory data and characterisation data were not reported, making assessment of these results difficult.

2.4.4 SUMMARY OF METHODOLOGICAL KNOWLEDGE GAPS AND ADVANCEMENTS FROM CONTEMPORARY AUSTRALIAN RESEARCH

A total of nine contemporary livestock LCA studies have been completed in Australia before or during the time in which the research in this thesis was completed, with three studies covering beef (one including lamb), four studies focused on lamb and one study of chicken meat. Of these, two projects covered more than three indicators, while two studies covered GHG emissions and water, and the remaining five studies investigated GHG emissions only. A total of nine CSF and
two research sites were included in the beef and lamb studies. All but one study applied economic allocation to handle co-production between wool and lamb, and two studies used economic allocation among different classes of livestock. One study also applied biophysical allocation using custom methods. For meat processing, application of mass allocation was also a limitation because this allocates substantial burdens to residual products. Economic allocation is the least favoured method in the ISO (ISO, 2006a; ISO, 2006b) hierarchy, suggesting further research is needed to investigate new methods for handling co-production on-farm and for processing.

When limitations regarding system boundaries were taken into account, only two beef CSF (the Arcadia valley farm) (Eady et al., 2011) and the Victorian farm in 2004 (Peters et al. 2010a) were found that used primary inventory data, studied cattle that were at the point of slaughter and had no evident limitations arising from system boundary choices. Both studies applied enteric methane estimation methods and GWP values that were inconsistent with later research, resulting in a slight overprediction of GHG emissions results. The beef study, based on idealised data from gross margins, provided indicative values for water and GHG emissions, but it was unclear what omissions may have existed because overheads were excluded, or because of the impact of ideal herd productivity factors and estimated water supply attributes rather than actual data.

With respect to wool and lamb, four studies covering six CSF (two covering GHG emissions only, three covering water only, and one covering GHG and water) were found that had no apparent system boundary limitations. The results for GHG emissions were broadly similar despite differences in allocation choices, which made comparison difficult. Water results were higher from the WA study compared to the Victorian study without clear explanation.

The one contemporary chicken meat study contributed meaningfully to some aspects of knowledge regarding impacts from chicken meat. However, the method choices, data quality for important supply chain stages such as grow-out, and model choices for predicting gaseous emissions produced results that were unlikely to be fully reflective of typical Australian production. These limitations were overcome in the study presented in chapter 8.
From this summary, the following important contributions were identified from the literature:

- Input-output hybrid models produce only small improvements in the comprehensiveness of LCA of red meat supply chains (Peters et al. 2010a) and this method is therefore less important for livestock systems than noted elsewhere in the literature. Nonetheless, input-output data were utilised to improve the comprehensiveness of the published studies incorporated in this thesis by including impacts associated with purchased services.

- Comparison of supply chains must be done with cattle of comparable weight or stage in the supply chain. Impacts from weaners are higher than slaughter cattle (Eady et al., 2011). This important conclusion was taken into account in the research of beef supply chains (chapter 3) which focused on production of cattle that were ready for slaughter to avoid unequal comparisons.

- Water assessment is very sensitive to method choices, and actual water abstractions are a small proportion of the total water balance on grazing farms (Peters et al. 2010b). New methods were adopted to determine net water requirements in the farm system, including impacts from farm dams.

- Water use methods for predicting drinking water and livestock water flows were provided by Ridoutt et al. (2012a) and these methods were adopted in the author’s study (chapter 3).

- The possible significance of evaporation from farm dams was identified by Ridoutt et al. (2012a) by calculating theoretical water balances, and this study also demonstrated the effect of applying water impact assessment methods. Evaporation from farm dams was assessed using a detailed modelling approach, using activity data collected from an on-farm assessment of water supply systems and historical farmer records across a large number of case study farms (reported in chapter 3, 4, 5 and 7).

- Eady et al. (2012) contributed an important investigation of biophysical methods for handling co-production on wool farms. These methods were used as a starting point for developing new biophysical allocation methods for wool production.
However, considering the very small number of CSF studied and the lack of studies using larger datasets, it can be concluded that insufficient information was available to benchmark GHG emissions, energy or water associated with beef, lamb, chicken or pork from the contemporary research. No studies performed uncertainty analyses using Monte Carlo analysis, though some studies assessed variability using scenario analysis or sensitivity testing. This left knowledge gaps regarding uncertainty in the results for all livestock species. These gaps were partly rectified by the published studies in this thesis, which substantially expanded the case studies and regional analyses for each meat production system, and included uncertainty analysis in most cases.

Additionally, knowledge gaps existed in all industries with respect to GHG emissions and water, because of omissions with respect to LU and dLUC emissions, and application of GHG emission prediction methods and water prediction methods that were subsequently superseded. These knowledge gaps were addressed by including LU and dLUC emissions in each study (in the case of beef, this was done via an associated study by the author and colleagues, cited as Henry et al. 2015). Updated GHG calculation methods and water prediction methods were applied in the published studies, or where necessary modifications were made to improve comparability of the data presented in the discussion. Key methods are described in the following section.

2.5 INITIAL STUDIES AND PUBLISHED STUDIES

This section provides research context for the subsequent chapters and discussion, and importantly, outlines how the initial studies contributed to objective iv. It examines the sensitivity of key assumptions relating to greenhouse gas emission prediction providing recommendations and guiding research to quantify and reduce these impacts.
Analysis of resources and GHG emissions from meat production

Figure 1. Research timeline showing the author’s research from 2009-2018
2.5.1 PORK AND CHICKEN MEAT

Two initial studies were completed for the pork and chicken meat industries in 2010 and 2012. The first was an assessment of pork supply chains, which was completed in two parts (Wiedemann et al., 2010b - see supplementary material) and an expanded pork LCA (Wiedemann et al., 2012a). This study covered GHG emissions, energy and water, including farm and meat processing, with the second part expanding the scope to include more case study farms and mitigation methods. The second was an assessment of two major meat chicken production regions (Wiedemann et al., 2012b - see supplementary material). These studies identified sensitive and uncertain emission factors regarding GHG emission prediction, potential mitigation strategies, gaps regarding the coverage of different production systems, and identified deficiencies in the water methods proposed by Harris and Narayanaswamy (2009a). New water assessment methods were proposed. Key sensitive GHG emission factors and the studies that were commissioned to validate these are summarised in Table 1.
Table 1. Sensitive and uncertain certain greenhouse gas emission factors identified during the initial research for pork and chicken meat

<table>
<thead>
<tr>
<th>Description</th>
<th>Emission factor applied in the NIR*</th>
<th>Significance and uncertainty in NIR factors</th>
<th>Research commissioned to validate or revise emission estimation</th>
</tr>
</thead>
</table>
| Pork – manure nitrogen (N) and volatile solids (VS) estimation techniques. | Fixed values applied for different classes of pigs. | • Initial research showed manure VS and N vary substantially between piggeries and with different levels of feed waste.  
• Contemporary research demonstrated high levels of variability in N and VS excretion (McGahan et al., 2010). | Parallel research by Skerman et al. (2015) |
| Pork – methane conversion factor (MCF) and ammonia factor for anaerobic ponds. | Open anaerobic pond MCF = 0.9, Ammonia N = 0.4. | • Sensitivity analysis factors found to change total GHG emissions by up to 35%.  
• European factors not verified with Australian research. | McGahan et al. (2016) |
| Pork – nitrous oxide factor for deep litter housing. | No factor applied in NIR. The IPCC recommended factor of 0.02 kg N₂O-N kg N⁻¹ excreted (Dong et al., 2006). | • Sensitivity analysis found to change total GHG emissions by up to 60%.  
• No factor applied in the NIR at the time of the research being commissioned (this type of housing was not identified).  
• LCA adopted European factors from the IPCC (Dong et al., 2006), not tested in Australian conditions. | Phillips et al. (2016) |
| Chicken meat – nitrous oxide and methane from manure (with bedding) | Default factors for feed intake and manure ash:0.02 kg N₂O-N kg N⁻¹ excreted, 1.5% MCF. | • Overestimated manure N excretion by 10%.  
• Sensitivity analysis factors found to change total GHG emissions by up to 19%.  
• Based on default values and not calculated via mass balance. Inconsistent with recommendations in the IPCC 2006 (Dong et al., 2006) and not verified with Australian research. | Wiedemann et al. (2016d) |

*NIR: National Inventory Report (DCCEE, 2010)
2.5.2  **BEEF, LAMB AND WOOL STUDIES**

One review and two scoping projects were completed for the beef and lamb industries in 2009 and 2010. A review of methodology issues and options to reduce GHG emissions and water in red meat was commissioned by Meat and Livestock Australia (MLA) in 2009 and completed in 2010. The report (Wiedemann et al., 2010a) and publications (Cottle et al., 2011; Wiedemann and McGahan, 2010) are provided as an electronic supplementary to this thesis. This review identified the need for common methods to handle co-products, common functional units (FU), system boundaries and specific GHG emissions and water calculation methods, together with recommendations around the need for differentiating arable and non-arable land. Concurrently, the author completed scoping LCAs for grain-fed and extensive beef (i.e. Wiedemann et al., 2010c - see supplementary files). These reports identified sensitive and uncertain emission factors regarding GHG emission prediction, summarised in Table 2.

### Table 2. Sensitive and uncertain certain greenhouse gas emission factors identified during the initial research for beef

<table>
<thead>
<tr>
<th>Description</th>
<th>Emission factor applied in NIR*</th>
<th>Significance and uncertainty in NIR factors</th>
<th>Research commissioned to validate or revise emission estimation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Feedlot beef – nitrous oxide and methane from the feed pad and stockpile</td>
<td>0.02 kg N₂O-N kg N⁻¹ excreted, MCF=1.5 or 2%</td>
<td>Contributed 65-82% of the manure GHG emissions. European factor adopted from the IPCC (1997). Climate and management conditions substantially different in Australia.</td>
<td>Redding et al. (2015b)</td>
</tr>
<tr>
<td>Feedlot beef – indirect nitrous oxide via atmospheric deposition of ammonia</td>
<td>0.01 kg N₂O-N kg NH₃-N⁻¹ lost, 0.3 kg NH₃-N kg N⁻¹ excreted</td>
<td>Contributed 9% of the GHG emissions from manure. European factors adopted from the IPCC (1997). Nitrous oxide factor may be higher than warranted in Australian conditions. Ammonia emission factor was lower than contemporary research and the National Pollutant Inventory (FSA, 2006).</td>
<td>Redding et al. (2017) &amp; Redding et al. (2016)</td>
</tr>
<tr>
<td>Manure application nitrous oxide</td>
<td>0.01 kg N₂O-N kg N⁻¹ applied</td>
<td>21% of feedlot manure GHG emissions. European factor adopted from the IPCC (1997). Climate and management conditions substantially different in Australia.</td>
<td>Limited work has been commissioned in this area focusing on inventory verification</td>
</tr>
<tr>
<td>Grass-fed beef – Enteric methane</td>
<td>Tropical pastures: Kurihara et al. (1999), Temperate pastures: Blaxter &amp; Clapperton (1965)</td>
<td>Enteric methane contributed &gt;90% of total GHG emissions in the initial studies for tropical cattle. 36% higher than IPCC default (DCCEE, 2010), not supported by later Australian research subsequently published as Kennedy &amp; Charmley (2012).</td>
<td>Work by CSIRO already underway to address this, i.e. Kennedy &amp; Charmley (2012).</td>
</tr>
</tbody>
</table>

*NIR: National Inventory Report (DCCEE, 2010)*
2.5.3 SUMMARY OF KEY OUTCOMES FROM THE INITIAL STUDIES

The initial studies provided the base data and assumptions for the published studies, with some additions or modifications (explained in the introduction and addenda for each paper).

1. **Revision of the NIR** (Commonwealth of Australia, 2016). Identification of sensitive GHG emission factors, manure methods and inventory methods in the LCA research led to a review, commissioned by the Federal Department of Environment and completed by the author, which recommended more than 95 new factors or methods for the inventory, of which almost half were omitted from the inventory prior to this date (see Wiedemann et al., 2014c in the supplementary material, and Commonwealth of Australia, 2016).

2. **GHG emission mitigation and validation research and development.** Research was commissioned in response to the sensitivity analysis and mitigation studies, resulting in > $8 M of commissioned research under the NAMMP program and direct industry commissioned research. Additionally:
   a. LCA research was used to establish industry targets for reducing GHG emissions in pork production to 1 kg of CO₂-e kg pork⁻¹ (liveweight), which formed a key goal and performance indicator of the Pork CRC (Co-operative Research Centre) and the Pork CRC biogas support program (Pork CRC, 2018), and
   b. Supporting development of the first method for mitigating GHG emissions under the Australian Carbon Farming Initiative (CFI) for the *Destruction of Methane Generated from Manure in Piggeries* (Commonwealth of Australia, 2015b). LCA research was used to demonstrate the calculation methods and leakage risks.

3. **Energy and water efficiency research.** The pork and meat chicken LCA studies identified hotspots and variability in water and energy results, directing subsequent research to determine efficiency opportunities in both industries (McGahan et al., 2014a; McGahan et al., 2014b).

4. **LCA method development.** UN FAO LEAP program – contribution to methods of Poultry (LEAP, 2014), Swine (LEAP, 2016), Large Ruminants (LEAP, 2015a) and Water Assessment (ongoing).
Significant methodological outcomes relating to the initial LCA research were as follows:

1. Water methods proposed by Harris and Narayanaswamy (2009a) were outdated by new research. The two most common approaches applied either water footprint methods (Hoekstra et al., 2009) which included analysis of green, blue and grey water, or new LCA methods (Milà i Canals et al., 2009; Pfister et al., 2009) which recommended excluding green and grey water from the impact assessment. Application of LCA water methods that excluded green water was recommended in these initial studies and was applied in the published studies, because of the difficulty in defining and communicating the impact of green water.

2. Initial research for the wool industry identified that methods used for handling co-production of wool and liveweight was a key limitation to harmonising LCA research in this sector. This was subsequently addressed by the study of Wiedemann et al. (2015b) “Application of life cycle assessment to sheep production systems: investigating co-production of wool and meat using case studies from major global producers” with a consortium of international researchers. The findings were applied in chapters 5, 6, and 7.

In this way the initial LCA research completed within the scope of this thesis addressed the key objectives of examining sensitivity of key GHG emission assumptions and guiding research.

2.5.4 Expansion of the project scope to include land occupation and HEPR

2.5.4.1 Land Occupation

The Australian literature studies and the author’s initial pork and chicken meat studies did not include land occupation in the scope of the study in all instances, with the exception of Bengtsson and Seddon (2013) who didn’t report inventory or characterisation results.
To address this knowledge gap, the published studies included an expansion of the inventory to assess land occupation in all industries, differentiating between crop land (sometimes termed arable land in the thesis chapters) and rangeland.

2.5.4.2 HUMAN EDIBLE PROTEIN CONVERSION

In addition to assessing environmental impacts and resource use, this study aimed to consider impacts and trade-offs that relate to food security. One potential trade-off between different meat products relates to the use of grain for meat production that could otherwise be used directly for human consumption. LCA typically assesses resource use (such as fossil fuel energy) throughout the system back to the point of extraction, but this does not account for the resource value or role of grain in global food security. To improve the knowledge in this respect, the mid-point indicator human edible protein required (HEPR) was applied to measure the proportion of protein required from human edible sources, relative to the human edible protein produced. HEPR was determined by calculating the human edible protein in the feed consumed per kilogram of human edible protein produced. Thus, numbers exceeding one (1) indicate the system consumes more human edible protein than it produces and therefore reduces human protein supply. Conversely, numbers below 1 indicate a net contribution to human protein supply. Note that chapter 6 used the term human edible protein conversion efficiency (HEP-CE) which is the inverse of HEPR (i.e. values indicate the amount of human edible protein produced from each kilogram of human edible protein consumed; values exceeding one (1) produced more human edible protein than they consume). This indicator is similar to the indicators reported in Gill et al. (2010). The published studies for chicken meat and pork did not include an analysis of HEPR, and methods used to determine HEPR for these studies were included in the addendum to each paper.
3 RESOURCE USE AND ENVIRONMENTAL IMPACTS FROM BEEF PRODUCTION IN EASTERN AUSTRALIA INVESTIGATED USING LIFE CYCLE ASSESSMENT

3.1 ATTRIBUTION STATEMENT

The paper Resource use and environmental impacts from beef production in eastern Australia investigated using life cycle assessment was led by S.G Wiedemann, and co-authored by E.J. McGahan, C.M. Murphy and M-J. Yan. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions
- Development and application of novel methods for determining water use from farm storages
- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and herd outputs
- Primary data acquisition and data analysis
- Preparation of the manuscript and completion of the peer-review process

McGahan contributed to the study and manuscript via:

- Valuable assistance to development of the on-farm water modelling methodology
- Assistance with data collection
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation

Yan contributed to the study and the manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation
3.2 INTRODUCTION

This paper integrated case study and regional analysis research covering a range of geographies in eastern Australia, utilising case studies completed in a series of LCA projects. This was the first ruminant case study paper developed by the author as part of the research covered by the thesis, and established important methodological approaches around modelling farm water use, land occupation and integrating case study and regional datasets. Four companion papers were also written, covering i) grain finishing systems (chapter 4), ii) the extended supply chain through to export markets (chapter 6), an analysis of land use and direct land use change impacts (Henry et al., 2015a) and analysis of the interactions between land use change and water use (Wiedemann, 2014).

The study advanced the concepts developed by Peters et al. (2010b) and Ridoutt et al. (2012a) regarding assessment of water use on grazing farms. The study quantified the assumption based work of Ridoutt et al. (2012a), showing that evaporation from farm dams was a source of substantially higher losses than shown by the theoretical assessment of Ridoutt et al. (2012a). The study developed new knowledge regarding dam efficiency (the only known assessment of its kind in Australia at the time), using daily time-step water balance modelling for dams on each case study farm, validated by comparing with farmer records of drying and overtopping events. These inventory methods have been integrated into LEAP guidance for assessing water in livestock systems.

The study filled an important knowledge gap with respect to crop land occupation in livestock systems, utilising satellite imagery and farm scale validation.

The study developed methods to integrate case study and regional datasets (based on ABARES datasets) to assess and compare regional influences, revealing new insights into water and energy use. One key finding from the regional assessment was the substantial contribution from irrigation to overall water use, though used on only a small number of farms.
The study was the first to adopt a uniform method for estimating enteric methane emissions between northern and southern regions following the early findings of Australian research, later published by Charmley et al. (2016). This removed the inaccuracies in comparing beef systems between these regions caused by the application of different emission estimation methods. Importantly, the dataset enabled development of a regression of GHG emissions and herd productivity factors, enabling a prediction method to be developed for herd GHG emissions per kilogram of beef for similar systems. This study applied a novel approach to handle co-production allocation within beef herds, advancing previous research in this area. Allocation of meat from different market types in a herd (young animals or older, cull animals) was avoided by considering these outputs functionally equivalent from a human nutrition perspective. This novel approach was adopted by international livestock LCA guidance (see LEAP, 2014; LEAP, 2015a; LEAP, 2016).
Resource use and environmental impacts from beef production in eastern Australia investigated using life cycle assessment

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Abstract. Resource use and environmental impacts are important factors relating to the sustainability of beef production in Australia. This study used life cycle assessment to investigate impacts from grass-finished beef production in eastern Australia to the farm gate, reporting impacts per kilogram of liveweight (LW) produced. Mean fossil fuel energy demand was found to vary from 5.6 to 8.4 MJ/kg LW, mean estimated fresh water consumption from 117.9 to 332.4 L/kg LW and crop land occupation from 0.3 to 6.4 m²/kg LW. Mean greenhouse gas emissions ranged from 10.6 to 12.4 kg CO₂-e/kg LW (excluding land use and direct land-use change emissions) and were not significantly different (P > 0.05) for export or domestic market classes. Enteric methane was the largest contributor to greenhouse gas emissions, and multiple linear regression analysis revealed that weaning rate and average daily gain explained 80% of the variability in supply chain greenhouse gas emissions. Fresh water consumption was found to vary significantly among individual farms depending on climate, farm water supply efficiency and the use of irrigation. The impact of water use was measured using the stress-weighted water use indicator, and ranged from 8.4 to 104.2 L H₂O-e/kg LW. The stress-weighted water use was influenced more by regional water stress than the volume of fresh water consumption. Land occupation was assessed with disaggregation of crop land, arable pasture land and non-arable land, which revealed that the majority of beef production utilised non-arable land that is unsuitable for most alternative food production systems.

Additional keywords: carbon, cattle, footprint, land, LCA, water.

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Introduction

The world’s population is expected to reach 9.1 billion people by 2050 (FAO 2009) and with a global population increasing in number and affluence, demand for meat is forecast to increase by 74% by 2050 (FAO 2009). To meet this demand will require agricultural industries to increase production significantly, but the impacts of this production on resources and other environmental impacts also need to be taken into account. Considering this, the efficiency of resource use and environmental impacts associated with producing beef is of growing importance. The beef industry was the largest agricultural contributor to Australia’s direct greenhouse gas (GHG) emissions in 2012 (Commonwealth of Australia 2014). With global targets in place to reduce GHG emissions (IPCC 2013) and a requirement to maintain or increase beef production to meet global food demands, further information is required on the emissions intensity of production. A limited number of studies (Peters et al. 2010a; Eady et al. 2011) have reported emissions intensity for Australian beef using the life cycle assessment (LCA) approach, which accounts for impacts across the whole supply chain. These studies reported two farms each, in two different regions. Differences in methodology and the application of different enteric methane prediction methods has limited the understanding of emissions intensity of beef from case studies and there is little information available to indicate the likely impacts across the industry because of the small sample sizes used.

With respect to Australian water resources, these are constrained in major river systems such as the Murray–Darling (ABS 2008a), and restricted (capped) supply has led to increased competition between water users (MDBA 2012). Water use by grazing industries is not well understood in Australia and is complicated by different water definitions (Wiedemann and McGahan 2010). Some authors have indicated the Virtual Water Content (VWC) for beef production may be very high, with estimates between 17 112 L/kg for Australian beef (Hoekstra and Chapagain 2007) and 120 000–200 000 L/kg beef (Pimentel et al. 2004). Such results are calculated using very different methods than the traditional understanding of water management as defined by the Australian Bureau of Statistics national water accounts (ABS 2010), where catchment-scale water balances are applied. The VWC method includes evapotranspiration losses associated with rain falling directly on pastures – so called ‘green water’ in addition to water from extracted sources such as rivers or dams, termed ‘blue water’
Resource use and impacts from beef

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(Falkenmark 2003). Mekonnen and Hoekstra (2012) identified that 96% of the water footprint from global beef production is green water, which is not considered a water resource in the traditional sense and is not part of Australia’s water accounts (i.e. ABS 2010). In contrast to the water footprint results, Peters et al. (2010b) determined water use using methods consistent with the ABS (2010) and found water use from beef production to be 27–540 L/kg carcass weight for two case study farms (CSF) in southern Australia. However, this study did not account for some on-farm water sources and water supply losses such as evaporation from farm dams, which is also understood to be an important water use in Australia (Nathan et al. 2004). Although Ridout et al. (2012b) applied new methods that included supply losses in their assessment of Australian beef water use, the study was based on theoretical case studies only, suggesting knowledge of water use on beef farms is still incomplete.

Other resources used by beef producers such as energy and land have received much less attention. Only one previous LCA study investigated energy use from beef production (Peters et al. 2010a) for two CSF in southern Australia and only one beef study has investigated land occupation. Across other livestock sectors, multiple impact studies of Australian production have been completed for lamb production (Wiedemann et al., 2015a), pork (Wiedemann et al., 2010), meat chickens (Wiedemann et al., 2012) and eggs (Wiedemann and McGahan, 2011). Grazing industries use 55.7% of Australia’s land resources (Lesslie and Mewett, 2013), though most of this is classified as ‘grazing native vegetation’ in rangeland areas. In contrast, only 3.5% of Australia’s land mass is used for dryland or irrigated crop production (Lesslie and Mewett, 2013). Crop production land is suited to a wide range of alternative production systems compared with non-arable rangeland areas and warrants consideration as a separate resource.

In the context of these impacts and constraints, there is a need for holistic analysis that investigates multiple impacts of beef production concurrently. To provide a holistic analysis that avoids shifting burdens from one impact to another or from one stage of the supply chain to another (burden shifting), a full supply chain approach such as LCA is required. LCA can also be used to assess multiple resource use and impact issues within the same research framework, presenting results per kilogram of product; an intensity metric, which is ideal for understanding and improving productivity against resource and environmental goals. This study aimed to investigate multiple resource and environmental impact issues from grass-fed beef production in two major eastern Australian production regions, using case studies and survey data to produce benchmark results and a hot-spot analysis of impacts. The study applied new methods for assessing water use, energy use and land occupation not previously reported for CSF in Australia, covering a broad range of climate zones and cattle produced to different market weights for either domestic or export markets.

Materials and methods

Description of production systems

This study focussed on major prime grass-fed beef production regions in eastern Australia and included the following Queensland (Qld) regions: part of the Mackay region, the

Fitzroy, Burnett-Mary, Darling Downs and Maranoa. In New South Wales (NSW) farms in the following regions were assessed: the North-West, New England, Central West and Southern Tablelands. Regions were selected to align with major grass-fed beef production areas that have the capacity to produce cattle at growth rates suitable for domestic and premium export grass-fed markets. These regions represent ~35% of Australia’s breeder herd (ABS 2012a) and are shown in Fig. 1. The study included 11 CSF, which were assessed individually with on-farm visits and surveys of production, combined with an analysis of regional average farms (RAF) modelled from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) survey of specialist beef producers (ABARES 2013). The regional analysis used data extracted for the 5-year period to 2010 for regions 322, 321 and 331 in Qld (Qld RAF – total survey of 191 farms), and regions 121, 122 and 131 in NSW (NSW RAF – total survey of 154 farms). The CSF were modelled separately and results were combined to provide a non-weighted mean. All but two CSF were located in the RAF regions, with the remaining two farms being included because they provided additional data for climatic conditions and production systems representative of the RAF regions. The systems were modelled through to the production of a slaughter ready animal, which included distinct production stages (breeding, backgrounding on pasture and finishing on pasture or forage) on some farms (Fig. 2). The systems were grouped on the basis of market destination to either domestic retail or export markets. In Australia, the domestic fresh meat market requires younger cattle <2 years of age at slaughter, with slaughter weights of typically <500 kg liveweight (LW), whereas export markets accept older cattle up to 2–4 years of age at slaughter and typically at >500 kg LW (Andrews and Littler, 2007). In many cases, farms sold cattle to both markets. In the regional assessments, grass-finished domestic cattle were only modelled in NSW where typical breed types and growth rates produce cattle

Fig. 1. System boundary for grass-finished beef production in eastern Australia.
more suited to this market compared with Qld. Market classes were divided in order to consider differences in the age and weight of finished cattle.

**Indicators, system boundary and functional unit**

The study investigated GHG emissions using IPCC AR4 global warming potential values (IPCC 2007), fossil fuel energy demand measured in MJ with lower heating values (Frischknecht et al. 2007), fresh water consumption and stress-weighted water use (Pfister et al. 2009) and land occupation. The study followed guidance from ISO (2006). The system boundary included all processes associated with the production of beef up to the farm gate. Impacts from direct land-use change (dLUC) were taken into account compared with a reference system 20 years before the analysis period (PAS 2050: 2011 – BSI 2011), which was taken as the year 1990. Impacts on GHG emissions resulting from dLUC are reported in a separate publication. The functional unit was chosen as ‘one kg of LW at the farm gate’.

**Life cycle inventory**

Detailed production data, livestock inventories and input data were collected from farm records, interviews and site visits to each CSF. Critical herd productivity factors such as weaning rates and sale weights were cross checked against 2–3 years of data to remove the effect of unusual seasons. Livestock growth rate assumptions for the RAF were based on those provided in the Australian National GHG Inventory (DCCEE 2012) and Bortolussi et al. (2005), in the absence of growth rate data in the ABARES survey. Dry matter intake for cattle grazing pasture was modelled from the herd inventory using the prediction equation of Minson and McDonald (1987). Herd characteristics are provided in Table 1 and farm characteristics are shown in Table 2.

Purchased inputs were summarized in Table 3. Energy demand was determined from purchased fossil fuels and electricity use, commodity use, feed purchases and farm services. Transport records were determined from farm data for livestock movements, purchased inputs and staff travel to and from work. Impacts associated with farm infrastructure were excluded based on the findings of a scoping study showing the contribution from these sources was <1% (data

![Fig. 2. Map of eastern Australia showing focus regions in Queensland (Qld) and New South Wales (NSW).](image)

| Table 1. Cattle production parameters of grass-finished beef from case study farms (CSF) and regional average farms (RAF) |
|---|---|---|---|---|---|
| | Queensland CSF | RAF | New South Wales CSF | RAF |
| Mean cow weight (kg LW) | 478 | 445–511 | 489 | 473 | 435–499 | 458 |
| Cows mated annually | 1343 | 212–3500 | 306 | 313 | 85–871 | 180 |
| Liveweight sold per year (t LW) | 522.5 | 82–1402.4 | 112.7 | 140 | 35–410.5 | 71.5 |
| Feed consumption (t DMI) | 10.532 | 1510–28253 | 2385 | 2409 | 603–6843 | 1451 |
| Weaning per cent (%) | 75.5 | 61.9–83.0 | 73.3 | 89.4 | 85.0–95.5 | 85.3 |
| Breeder culling rate (%) | 20.7 | 16.0–25.1 | 20.0 | 15.2 | 13.0–20.2 | 15.0 |
| Mortality rate (%) | 2.4 | 1.5–4.0 | 1.8 | 1.6 | 1.5–2.0 | 2.3 |
| Weaning weight (kg LW) | 220 | 153–275 | 216 | 225 | 209–240 | 205 |
| Weaning age (months) | 7.2 | 6.0–8.0 | 7.0 | 8.0 | 7.5–8.2 | 8.0 |
| Export steer sale weight (kg LW) | 581 | 542–610 | 549 | 597 | 537–704 | 557 |
| Export steer ADG (kg/day) | 0.57 | 0.45–0.67 | 0.55 | 0.64 | 0.54–0.76 | 0.52 |
| Domestic steer/heifer sale weight (kg LW) | 451 | 440–462 | n.a | 424 | 415–431 | 428 |
| Domestic steer/heifer ADG (kg/day) | 0.56 | 0.55–0.59 | n.a | 0.71 | 0.63–0.76 | 0.59 |
Impacts were modelled based on expenditure using economic input-output data (Rebitzer et al. 2002) when associated with services such as communications, insurance and accounting, using USA input-output tables available in SimaPro 7.3 (Pré-Consultants 2012) in lieu of readily available Australian input-output data. The ABARES survey reported aggregated farm expenditure on purchases in broad categories and input data were required at a greater resolution for several inputs, requiring additional data and model assumptions described here. Within expenditure classes (i.e. fertiliser, fodder), the proportions and type of purchased products were determined from the CSF dataset, and product mass was calculated from expenditure using market values. These assumptions are provided in the supplementary material. Uncertainty associated with purchased inputs was determined using a pedigree matrix (Frischknecht et al. 2005), which took into account uncertainty in the disaggregation methods. Background data were sourced from the Australian LCI database (Life Cycle Strategies 2007) where available, or the European Ecoinvent (2.0) database (Frischknecht et al. 2005).

### Table 2. Land and water resources for the case study farms (CSF) and regional average farms (RAF)

<table>
<thead>
<tr>
<th></th>
<th>Queensland</th>
<th></th>
<th>New South Wales</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>CSF Export</td>
<td>CSF Domestic</td>
<td>RAF Export</td>
<td>RAF Domestic</td>
</tr>
<tr>
<td>Average annual rainfall (mm)</td>
<td>649 689 662</td>
<td>763 763 667 667</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Farm size (ha)</td>
<td>19 570 5296 5193</td>
<td>1152 1152 1770 1770</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pasture (ha)</td>
<td>19446 5092 5055</td>
<td>1062 1062 1417 1417</td>
<td></td>
<td></td>
</tr>
<tr>
<td>On-farm crop/forage (ha)</td>
<td>4 9 139</td>
<td>43 43 354 354</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Proportion of farmed area utilised for</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sheep (%)</td>
<td>0 0 1</td>
<td>36 36 17 17</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cattle (%)</td>
<td>100 100 96</td>
<td>60 60 63 63</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Cropping (%)</td>
<td>0 0 3</td>
<td>4 4 20 20</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Farm dam (%)</td>
<td>45 42 45</td>
<td>68 68 60 60</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Bore (%)</td>
<td>36 36 38</td>
<td>18 18 23 23</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Creek (%)</td>
<td>19 22 17</td>
<td>14 14 17 17</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Water stress index</td>
<td>0.16 0.01 0.34</td>
<td>0.18 0.18 0.02 0.02</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

A The Queensland farms were situated in the Moura, Mundubbera, Marlborough, Mitchell and Texas regions. One additional farm from the Charters Towers region was included to represent northern inland production systems with lower productivity.

B The New South Wales farms were situated in the Tenterfield, Walcha, Uralla and Wagga Wagga regions. One additional farm from Inverloch (Victoria) was included as a proxy for farms in the New South Wales south coast region.

### Table 3. Farm services reported per tonne of dry matter intake (DMI)

<table>
<thead>
<tr>
<th></th>
<th>CSF Exam</th>
<th>CSF Domestic</th>
<th>RAF Exam</th>
<th>RAF Domestic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Export</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Energy</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Electricity (kWh/t DMI)</td>
<td>1.38 1.35 2.10</td>
<td>3.38 3.32 3.16 3.16</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diesel (L/t DMI)</td>
<td>2.68 2.69 3.82</td>
<td>1.59 1.41 4.79 4.79</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Petrol (L/t DMI)</td>
<td>0.11 0.13 0.44</td>
<td>0.85 0.82 0.74 0.74</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Purchased feed</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feed supplements (kg/t DMI)</td>
<td>5.76 6.04 6.17</td>
<td>0.03 0.04 0.79 0.79</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hay (kg/t DMI)</td>
<td>1.40 1.67 22.22</td>
<td>12.93 15.52 19.98 19.98</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Grain (kg/t DMI)</td>
<td>0.00 0.00 4.94</td>
<td>7.73 9.27 4.76 4.76</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Forage crops (kg/t DMI)</td>
<td>7.92 15.84 0.00</td>
<td>22.37 27.97 0.00 0.00</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Fertiliser</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Super phosphate (kg/t DMI)</td>
<td>0.00 0.00 0.00</td>
<td>7.14 8.56 3.07 3.07</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Other fertilisers (e.g. potassium) (kg/t DMI)</td>
<td>0.10 0.19 0.00</td>
<td>3.69 4.43 0.00 0.00</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Soil conditioners (kg/t DMI)</td>
<td>0.00 0.00 0.00</td>
<td>39.35 47.22 0.00 0.00</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Other inputs and services</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Veterinary products ($/t DMI)</td>
<td>1.54 1.28 2.18</td>
<td>2.82 2.89 3.59 3.59</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Purchased cereal grain inventory data were from the dataset reported in Wiedemann et al. (2010) and Wiedemann and McGahan (2011).

**Fresh water consumption**

The study focused on fresh water consumption using comprehensive water balance methods (Bayart et al. 2010) to assess uses and losses throughout the foreground and background system. Fresh water consumption refers to evaporative uses or uses that incorporate water into a product that is not subsequently released back into the same river catchment (ISO 2014). It has been recommended that impacts on fresh water consumption as a result of dLUC are included in LCA (Milà i Canals et al. 2009; ISO 2014). However, operational models for accounting for impacts at the local and regional scales have not been developed in Australia, and conflicting processes exist. At the local scale, historic dLUC from forest to pasture will result in increased water yield across the CSF (author’s modelling, data not shown) following well established relationships (Brown et al. 2005). Investigating the role of dLUC in a reforestation scenario, Wiedemann (2014) showed that reforestation would reduce water yield, leading to much greater fresh water consumption where this occurred on livestock farms. However, at the regional scale, deforestation may induce meteorological changes that result in reduced precipitation and water yield (Nobre et al. 1991; Kanae et al. 2001; Pitman et al. 2004). Considering this conflict, further research is required to characterise the impacts of dLUC at local and regional scales to produce meaningful results. In the present study, assessment of water yield compared with the reference system of 1990 to provide consistency with the GHG results. This resulted in no predicted changes to water yield for the CSF datasets because dLUC occurred in an earlier time period, though impacts may still be relevant for the RAF analysis. For the RAF analysis, insufficient data and modelling capability exist to determine the magnitude or direction of change in water yield as a result of dLUC, and this was therefore assumed to be zero. Results using a zero water yield comparison point align with previous research (i.e. Ridoutt et al. 2012b) and the national water accounts (i.e. ABS 2012b).

Water degradation was not assessed because of a lack of regional characterisation and nutrient transport factors for grazing systems across the regions assessed.

Primary sources of fresh water consumption were livestock drinking water, drinking water supply losses and irrigation. Drinking water was supplied from bores, creeks and rivers, or farm dams (Table 2). The proportion of supply from each source was determined from the farm questionnaire, and was verified by an analysis of water supply points across each farm using satellite or aerial photograph imagery. Regionally averaged water supply data from the CSF dataset were taken as a proxy for the RAF in the absence of water supply data from the ABARES dataset.

Where losses associated with the supply of water were caused by the production system, they were attributed to livestock production. Losses from farm reticulation systems were determined from sources of leakage and evaporation from open tanks and troughs. Evaporation losses from creeks and rivers were endemic to the natural system and were not attributed to livestock. Farm dam water balances were constructed from the inflow, extraction rates, predicted evaporation and seepage using a daily time-step water balance over a 70-year period, using long-term rainfall and evaporation data (Jeffrey et al. 2001; DSITIA 2013). Catchment runoff (dam inflow) was modelled using USDA-SCS KII curve numbers (USDA NRCS 2007) with appropriate values determined from site observations of soil type, farming practices and farmer knowledge of the frequency of runoff events. Extractions of livestock drinking water were determined from the livestock inventory and the livestock watered from each water point. Drinking water for grazing cattle was predicted from feed intake, climate and feed characteristics for each farm using equations from Ridoutt et al. (2012b). Evaporation from the dam surface was estimated from Class A pan evaporation using a pan-factor of 0.75–0.9 (Burman and Pochop 1994; Craig 2006) and seepage losses were determined based on farmer records for isolated dams where losses were evident. Total losses were calibrated using records of filling and emptying events collected from the farmers. Seepage losses were evident only in some poorly constructed dams, where losses typically resulted in a soak below the dam that promoted grass growth. These losses were therefore considered a consumptive use analogous to irrigation, rather than a transfer to surface or groundwater.

Supply efficiency from farm dams is reported here as a fraction of water interception from the environment (see Table 4). Supply efficiency was influenced primarily by net evaporation, dam density (total volume stored per hectare) and surface area to volume ratio (see Table 4). Dam supply efficiencies of 0.21 (Qld) and 0.29 (NSW) were applied for the

<p>| Table 4. Climate and farm dam characteristics on beef case study farms (CSF) in Queensland and New South Wales |
|-----------------|--------------|------------------|-----------------|------------------|------------------|------------------|</p>
<table>
<thead>
<tr>
<th></th>
<th>Queensland CSF Export</th>
<th>Queensland CSF Domestic</th>
<th>New South Wales CSF Export</th>
<th>New South Wales CSF Domestic</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dam density (ML/km²)</td>
<td>2.8</td>
<td>2.7</td>
<td>11.1</td>
<td>11.1</td>
</tr>
<tr>
<td>Dam demand factorA</td>
<td>0.075</td>
<td>0.075</td>
<td>0.070</td>
<td>0.070</td>
</tr>
<tr>
<td>Dam surface area to storage volume (m²/ML)</td>
<td>344.7</td>
<td>394.0</td>
<td>484.9</td>
<td>484.9</td>
</tr>
<tr>
<td>Dam supply efficiency</td>
<td>0.21</td>
<td>0.20</td>
<td>0.34</td>
<td>0.34</td>
</tr>
<tr>
<td>Avg. annual rainfall (mm)</td>
<td>667.5</td>
<td>731.3</td>
<td>780.5</td>
<td>780.5</td>
</tr>
<tr>
<td>Avg. annual evaporation (mm)</td>
<td>1972.7</td>
<td>1826.3</td>
<td>1327.6</td>
<td>1327.6</td>
</tr>
<tr>
<td>Net evaporation (mm)</td>
<td>1305.2</td>
<td>1095.0</td>
<td>547.1</td>
<td>547.1</td>
</tr>
</tbody>
</table>

A Volume of water extracted annually as a fraction of total water volume.
RAF datasets, derived from the non-weighted means from the CSF dataset in each region. Irrigation water use was contributed by both on-farm sources and off-farm sources via purchased feed. On-farm irrigation was only used on one case farm, and extraction and irrigation rates were available to determine total use at this farm. The ABARES survey provided an inventory of average irrigation land occupation. We determined total irrigation by applying an average irrigation rate for beef cattle pasture reported by the ABS survey of water use on Australian farms (ABS 2008b, 2012c). Uncertainties exist in this analysis, particularly with regard to irrigated land areas from the ABARES dataset. To reduce this uncertainty, total estimated irrigation water use was verified by comparison with reported total irrigation for ‘meat cattle’ for the years 2007/8–2009/10 (ABS 2008b, 2012c). Irrigation associated with feed grain inputs was modelled using the water inventory of Ridoutt and Poulton (2009). Losses associated with irrigation water supply of 27.1% were applied based on the ABS national water accounts (ABS 2012b).

Stress-weighted water use was determined by multiplying the total fresh water consumption in each region with the appropriate water stress index (WSI) values from Pfister et al. (2009). The value was then divided by the global average WSI (0.602) and expressed as water equivalents (H2O-eq; Ridoutt and Pfister 2010).

Land occupation

Land occupation was determined using a disaggregated land inventory that accounted for differences in land capability. We used three categories: crop land (arable land used to produce crops), arable land used for pasture (land suitable for crop production currently used for pasture, termed arable pasture land) and non-arable land, which is unsuitable for crop production because of limitations to land capability. These broad classifications provided an indication of the degree of disturbance associated with land occupation and the value of the land for alternative agricultural uses. The inventory of land occupation by land capability (Table 2) was determined from farm visits and discussions with farm owners, and was verified using aerial photography and satellite imagery of the CSF farms. Arable land in the RAF analysis was modelled from the inventory of grain and hay inputs. Total pasture land occupation was reported in the ABARES dataset but land capability was not detailed. In lieu of regionally specific data, arable pasture land was determined from the estimated area of crop land and total arable land as a proportion of grazing land at the national scale (4.9%), based on an analysis of data from the FAO (FAOSTAT 2014) and Lesslie and Mewett (2013). Further details are provided in the supplementary material. No characterisation factors were applied, and land occupation data were reported in square metre years (m² year).

**Greenhouse gas emissions**

Greenhouse gas modelling followed Australian (DCCEE 2012) or global inventory methods (IPCC – Dong et al. 2006) and are specified in Table 5. Australian methods for the prediction of enteric methane from grazing ruminants currently apply different methods for northern and southern beef cattle with substantially different results. Enteric methane prediction models from recent research in northern Australia (Kennedy and Charmley 2012) suggest emissions may be much lower than those measured by Kurihara et al. (1999) modified by (Hunter 2007), as applied by the Australian inventory. Australian inventory methods are currently being reviewed to address these differences (P. Reyenga, pers. comm.). To reduce uncertainty related to the choice of enteric methane model, we applied the IPCC method, which assumes enteric methane is 6.5% of gross energy intake. Uncertainty associated with the choice of emission factor was based on the IPCC (Dong et al. 2006). Uncertainty related to the prediction of feed intake was taken into account using an uncertainty range of ± 20%, based on the review by Poppi (1996). GHG emissions also arise from savannah burning, which was practiced on some farms. Fire management on the CSF focussed on controlled burning early in the season to maximise pasture growth. Emissions from controlled burning were attributed to the livestock system only where these exceeded emissions endemic to the reference system as a result of wildfires. The Australian GHG inventory indicates that managed fires replace wildfires that would naturally occur at other times of the year (DCCEE 2012). Early season burning was most common where fire management was practiced, and was expected to result in fewer emissions as a result of lower fuel loads than uncontrolled wildfires. As a result, no emissions were attributed to livestock from this source. GHG emissions or removals associated with soil carbon

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Key parameters/model</th>
<th>Uncertainty (%)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enteric methane (CH4 yield – % of GEI&lt;sup&gt;a&lt;/sup&gt;)</td>
<td>6.5% &lt;br&gt;kg DMI × (1 – DMD) × MEF&lt;sup&gt;b&lt;/sup&gt;</td>
<td>±5</td>
<td>IPCC – Dong et al. (2006)</td>
</tr>
<tr>
<td>Manure methane (grazing) (kg)</td>
<td>Urinary N – 0.004 kg N₂O-N/kg N in urine</td>
<td>±20</td>
<td>DCCEE (2012)</td>
</tr>
<tr>
<td>Manure nitrous oxide (grazing)</td>
<td>Faecal N – 0.005 kg N₂O-N/kg N in faeces</td>
<td>±50</td>
<td>DCCEE (2012)</td>
</tr>
<tr>
<td>Manure ammonia (grazing)</td>
<td>0.2 kg NH₃-N/kg N of excreted in manure</td>
<td>±20</td>
<td>DCCEE (2012)</td>
</tr>
<tr>
<td>Indirect nitrous oxide – ammonia</td>
<td>0.01 kg N₂O – N/kg N lost as ammonia-N</td>
<td>±50</td>
<td>DCCEE (2012)</td>
</tr>
<tr>
<td>Indirect nitrous oxide – leaching and runoff</td>
<td>0.0125 kg N₂O – N/kg N lost in leaching and runoff</td>
<td>±50</td>
<td>Applied regionally following the DCCEE (2012)</td>
</tr>
</tbody>
</table>

<sup>a</sup>Gross energy intake (GEI) assumed to be 18.6 MJ/kg DMI.

<sup>b</sup>Methane emission factors (MEF) [temperate (21°C) – 1.4 × 10⁻⁵ kg CH₄/kg DM manure, warm – 5.4 × 10⁻⁵ kg CH₄/kg DM manure].
change under pastures are known to be variable (Allen et al. 2013; Davy and Koen 2013; Wilson and Lonergan 2013). Considering this uncertainty in the magnitude and direction of soil carbon change under pasture, we assumed no change in soil carbon levels in the present study. Emissions from dLUC were assessed and will be reported in a later publication. Emissions from nitrogen fertiliser and lime application were included using methods outlined in the DCCEE (2012).

Handling co-production
There were several points in the production system where co-products were produced, and a method is required to divide burdens between products. In some cases, beef farms produced sheep or grain in addition to beef. Impacts from the co-production of beef, sheep and cereal grain were accounted for by subdividing the farm into systems and separating the unique impacts associated with each. For minor inputs that were shared by all production subsystems such as overheads, these were divided on the basis of land occupation. Land occupation by cattle and sheep was determined from the utilisation of feed resources by each species relative to the total annual feed production, and was used as a basis for apportioning inputs associated with pasture production. On farms where young cattle were produced for two or more market types, potential differences in efficiency between different market specifications were modelled by dividing the beef herd into separate, smaller herds, including all breeding and finishing animals, associated with each market type. Inputs and impacts were divided across the smaller herds based on relative feed intake, and unique inputs were attributed separately.

The beef system may co-produce beef destined for several markets from both young cattle and older cull breeding animals. We saw no clear rationale for allocating impacts separately to LW from young cattle and cull breeding animals, as meat products from all classes of cattle are suitable for human consumption and functionally comparable as a source of human nutrition. Thus within market classes, we attributed impacts evenly on a mass basis between all beef products.

Statistical analyses
Modelling was carried out using SimaPro 7.3 (Pré-Consultants 2012). Model uncertainty was assessed using Monte Carlo analysis in SimaPro 7.3. One-thousand iterations provided a 95% confidence interval for the results. Comparison of regions, datasets and market types was done using comparative Monte Carlo analysis in SimaPro 7.3 to remove the effect of shared uncertainty using methods described in Goedkoop et al. (2010). Multiple linear regression analysis was used to describe the influence of key herd parameters and farm variables on GHG emissions and fresh water consumption.

Results
Freshwater consumption and stress-weighted water use
Mean estimated Qld fresh water consumption ranged from 260.7 to 332.4 L/kg LW, which was significantly higher (P < 0.05) than fresh water consumption from the NSW production systems (see Table 6). Supply losses were the largest source of fresh water consumption in both regions, with a mean contribution of 59% in Qld and 45% in NSW. The mean contribution of drinking water and irrigation was 34% and 12% of total fresh water consumption across both regions for the RAF dataset (Fig. 3). Irrigation was found to be highly variable, ranging from 0 to 483 L/kg LW across the case farms. Irrigation water use was higher from export cattle than domestic cattle in the NSW CSF dataset, because export cattle from one farm were finished on irrigated forage. Acknowledging the small CSF dataset, irrigation water use is expected to be more representative from the RAF analysis where the contribution was 17.3–37.3 L/kg LW. When irrigation water use was removed, the regression model of fresh water consumption was:

\[
\text{Fresh water consumption (L/kg LW)} = -390 + 21.7\text{Temp} + 156.2\text{Dam}_F - 119.4\text{Dam}_SE; \quad R^2 = 0.78
\]

where: Temp is the mean maximum temperature, Dam_F is the fraction of total water supplied from dams compared with

### Table 6. Resource use and environmental impacts per kg of liveweight (LW) from case study farms (CSF) and regional average farms (RAF) in Queensland and New South Wales, reported for grass-fed export and domestic weight cattle

<table>
<thead>
<tr>
<th>Region and dataset</th>
<th>Number of farms</th>
<th>Market</th>
<th>Impact/inventory categories</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td></td>
<td>Freshwater consumption (L)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>Mean Range</td>
</tr>
<tr>
<td><strong>Queensland</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CSF</td>
<td>6</td>
<td>Export</td>
<td>281.3a</td>
</tr>
<tr>
<td>CSF</td>
<td>3³️</td>
<td>Domestic</td>
<td>260.7a</td>
</tr>
<tr>
<td>RAF</td>
<td>191</td>
<td>Export</td>
<td>332.4a</td>
</tr>
<tr>
<td><strong>New South Wales</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>CSF</td>
<td>5</td>
<td>Export</td>
<td>201.1b</td>
</tr>
<tr>
<td>CSF</td>
<td>5³️</td>
<td>Domestic</td>
<td>117.9c</td>
</tr>
<tr>
<td>RAF</td>
<td>154</td>
<td>Export</td>
<td>207.3b</td>
</tr>
<tr>
<td>RAF</td>
<td>154</td>
<td>Domestic</td>
<td>196b</td>
</tr>
</tbody>
</table>

³️ Number of farms also producing cattle for the domestic market.
other sources, and Dam SE is the supply efficiency of farm dams, expressed as a fraction.

Mean stress-weighted water use for Qld beef production was 8.4, 68.4 and 104.2 L H₂O-eq/kg LW from the CSF domestic, CSF export and RAF datasets, respectively (see Table 6) whereas mean values from NSW beef production ranged from 15.6 to 45.1 L H₂O-eq/kg LW. Queensland export beef used significantly more stress-weighted water ($P < 0.05$) than NSW export beef in both the RAF and CSF analysis. Stress-weighted water use was more closely related to regional differences in the WSI than to fresh water consumption. For example, Qld CSF domestic beef used more water in volumetric terms than NSW CSF domestic beef, but stress-weighted water use was higher from the latter, because a higher proportion of water was drawn from water-stressed sources.

### Land occupation

Mean crop land occupation ranged from 0.3 to 1.7 m²/kg LW in the Qld datasets and from 1.4 to 6.4 m²/kg LW in the NSW datasets. Crop land was significantly higher ($P < 0.05$) for the NSW export beef compared with Qld export beef, though differences were not evident between the domestic market types or the RAF datasets. Arable pasture land occupation was higher than crop land in both regions (see Table 7), and total land occupation was significantly higher in Qld because of the lower stocking rates and less intensive production compared with the NSW regions.
Energy demand

Mean fossil fuel energy demand was 5.6–8.4 MJ/kg LW (Table 6). Energy demand was significantly lower ($P < 0.05$) for Qld export cattle compared with NSW in both the CSF and RAF datasets. This trend was not evident between regions for the domestic market type. Lower energy demand from the Qld export beef production systems corresponded to lower inputs of supplementary feed, forage crops and fertiliser compared with the NSW farms. Among individual CSF, energy demand ranged from 3.9 to 12.5 MJ/kg LW with most of the variation caused by differences in farm fuel use, fertiliser use and supplementary feeding. Farm fuel use tended to be higher on Qld farms whereas fertiliser and supplementary feed use were higher in southern regions where production intensity and stocking rate were highest.

Greenhouse gas emissions

The mean estimated GHG intensity (excluding land use and dLUC) ranged from 11.9 to 12.4 kg CO$_2$e/kg LW in the Qld case study and regional analysis, and 10.6 to 12.2 kg CO$_2$e/kg LW in the NSW case study and regional analysis. Emissions were significantly lower ($P < 0.05$) from the NSW CSF compared with the Qld CSF or RAF. No significant differences were observed between export and domestic beef production. Contributions by gas types were similar among all the farms. Enteric methane was the single largest emission source, averaging 83% to 90%, followed by manure nitrous oxide (averaging 7% to 13%) and carbon dioxide from fossil fuels (averaging 2% to 9%).

The multiple linear regression analysis showed weaning percentage (Wean%) and average daily gain (ADG) were able to explain 0.33 and 0.47 of the variation in GHG intensity. The addition of mean pasture crude protein (CP) accounted for a further 0.07 of variation. The regression model of GHG emissions intensity was:

$$\text{GHG} = 20.73 - 0.047\text{Wean}\% - 11.06\text{ADG} + 0.13\text{CP}$$

($R^2 = 0.87$)

Discussion

This study applied a novel approach to assessing resource use and GHG emissions by integrating a small dataset from actual farms with an analysis based on regional survey data. This approach differed from previous case studies with one or two farms (Peters et al. 2010a, 2010b, 2011; Eady et al. 2011) or to studies using hypothetical data (Ridoutt et al. 2012a, 2012b) by providing both detailed site assessment for impacts and improved representativeness from the regional analysis. Specifically, the CSF dataset provided a detailed analysis of water supply factors, greater resolution in the land capability assessment, and a detailed inventory of purchased inputs not available in the RAF dataset. The much larger RAF dataset improved the representativeness of the inventory of purchased inputs and important inputs such as irrigation, which are only used by a small fraction of the industry but can have a large impact on results. Differences between the datasets were most obvious in the analysis of water and land occupation in NSW and this is discussed below.

Fresh water consumption and stress-weighted water use

The volume of fresh water consumption on grazing farms is an important factor in regional water resource management in Australia (Nathan et al. 2004; Cetin et al. 2009; Nathan and Lowe 2012). This study used a detailed water balance modelling approach and on-farm verification to estimate fresh water consumption and losses from farm dams, which has not been previously performed for case study LCA (i.e. Peters et al. 2010b; Eady et al. 2011). However, the analysis of irrigation water use was variable in the CSF analysis, as seen in the difference between the NSW export and domestic market classes. The higher water use in the export market class was the result of irrigation water use for forage crop production on one farm in the dataset, which was not practiced by any of the domestic beef producers. Irrigation water use was considered more representative in the regional analysis because of the larger dataset. Inclusion of a larger number of water-use sources and losses provided a more comprehensive analysis of water use for cattle production in the regions assessed, and suggested that mean fresh water consumption may be higher than previously reported in case study LCA (Peters et al. 2010b; Eady et al. 2011; Ridoutt et al. 2012b). However, fresh water consumption in these regions was lower than the national average reported by Wiedemann et al. (2015b), largely in response to lower irrigation water use in these regions compared with the national average, and lower supply losses from some water sources. Climatic factors had the greatest influence on fresh water consumption. Temperature and evaporation govern both livestock drinking water requirements and dam evaporation rates and both were higher in the Qld regions. Drinking water is an essential input that can not be readily reduced. However, more opportunities may exist to improve the efficiency of water supply, which was one of the primary factors governing fresh water consumption in this study. This was influenced by farm dam density, evaporative surface area and demand factors. Mean dam densities were 2.8 and 11.1 ML/km$^2$ for the Qld and NSW regions, respectively, which were similar to values reported by Nathan and Lowe (2012) of <3 ML/km$^2$ for southern Qld catchments and >10 ML/km$^2$ for several NSW regions corresponding to those assessed here. In contrast, dam demand factors were considerably lower for the CSF assessed here (mean: 0.07–0.075 and range of 0.03–0.11) than the 0.5 factor assumed by Nathan and Lowe (2012). Demand factors are an important measure of dam utilisation, with higher annual utilisation resulting in less surface area available for evaporation and higher supply efficiency. The demand factor of 0.5 reported by Nathan and Lowe (2012) implies farmers use 50% of the storage volume annually, which in addition to evaporation would result in dams emptying almost every year in many regions. Only one case farm out of the 11 reported that a proportion of dams on their farm emptied annually. This farm had access to bore and river water to maintain supply throughout the year reducing the reliance on dam water. Most farmers reported that they had 2–3 years of dam water storage without runoff events and water storage was particularly high on farms that did not have permanent alternative water supplies. Despite...
lower dam densities in the Qld farms, the mean supply efficiencies were still lower in Qld (0.2–0.21) compared with NSW (0.34) because of the higher mean net evaporation rate.

The fresh water consumption results were generated using a water balance approach at the farm level, which is similar to the water balance methods applied at the catchment level by the National Water Accounts (i.e. ABS 2012b) and therefore align with a traditional understanding of water resources. In contrast, alternative methods for assessing water use such as VWC attribute large volumes of ‘green’ water that is derived directly from rainfall to beef. For example, Mekonnen and Hoekstra (2012), report values $>17,000$ L/kg beef but of this, $<4\%$ is ‘blue’ water from sources such as rivers, bores or local water storage dams. It is incorrect to interpret such data as having any direct correlation with water stress or environmental impact.

Between case studies, regional water stress was found to have more influence on the impact of water use than the total volume used suggesting the location of water use is a more important factor governing impact on the environment than volume used. Stress-weighted water use in the NSW CSF dataset was influenced by one farm located in a highly water-stressed catchment, which resulted in an elevated mean WSI for the CSF compared with the RAF (Table 2), making results from the RAF analysis more representative of these regions. The range in stress-weighted water use across the CSF farms, from 2.0 to 361.7 L H$2$O-eq/kg LW, was broader than the 3.3 L–221 L H$2$O-e/kg LW reported by Ridoutt et al. (2012b) for an analysis of six theoretical production systems in NSW. The difference in the range of values related primarily to differences in the fresh water consumption inventory as the range of WSI values was similar in both studies.

**Land occupation**

This study is the first LCA known to the authors where land occupation for beef production has been disaggregated to report arable land and non-arable land separately.

In the two beef systems examined, only small amounts of arable land, and much larger proportions of non-arable rangelands were utilised. Availability of arable land is a significant global resource issue (UNEP 2014). Arable land can be utilised for a greater range of food production systems and can underpin production of alternative meat systems such as pork or poultry, which rely on grain inputs. Across many of the case studies, livestock were grazed on natural pasture lands without significant modification, with much lower disturbance than cultivation and more opportunity for high levels of natural biodiversity. Considering the very different characteristics of arable and non-arable land, comparison between beef and other livestock products on the basis of total land occupation (de Vries and de Boer 2010) would be more meaningful if these land types were considered as separate resources.

**Energy demand**

Mean energy demand was lower in the extensive production regions of Qld, though a wide range in energy demand was found across the individual case studies. Mean direct energy demand from the use of diesel, petrol, electricity and gas on-farm contributed 58% of total energy demand and was dominated by diesel use for farm operations. Indirect farm energy use was proportionally higher in NSW than Qld, reflecting higher inputs of fertiliser in NSW.

**Greenhouse gas emissions intensity**

Greenhouse gas emissions from livestock were found to be similar across market types. Emissions intensity was significantly lower for beef produced in NSW (CSF), as a result of better weaning percentages and growth rates in this region, though this trend was not observed in the RAF export analysis. Studies have previously demonstrated the relationship between herd productivity factors and methane emissions intensity for beef cattle (Hunter and Nieth 2009). This study extended the findings to show that full system emissions, including manure, crop nitrous oxide and carbon dioxide from fossil fuel use, could largely be explained by herd productivity factors. Analysis of the CSF dataset showed that higher weaning and growth rates reduced GHG even when this corresponded to higher inputs of supplementary feed or fertiliser. We note that the RAF analysis relied on assumed growth rates taken from surveys by Bortolussi et al. (2005) and assumptions from the DCCEE (2012) used in the National GHG Inventory. Considering the importance of these factors for predicting livestock GHG across industry and for the national inventory, expansion of the ABARES and ABS surveys to include weight and age of slaughter cattle would be beneficial. The growth rate assumptions for the NSW RAF analysis were lower than measured data from the CSF farms, resulting in higher emissions than the CSF and previous LCA of southern beef production (Peters et al. 2010a; Ridoutt et al. 2012a). Impacts from both the Qld and NSW RAF analysis were slightly lower than the average reported for the national beef herd by Wiedemann et al. (2015b), because these regions had slightly better productivity performance than the national average.

We found no relationship between finished weight and emissions intensity. Eady et al. (2011) found that weaned calves had higher emissions intensity than heavy finished steers, suggesting a relationship between animal LW and emissions intensity because of the ‘dilution of maintenance’ effect (Johnson et al. 1996). Our results suggest this effect has diminished by the time cattle reach marketable weights. Further investigation may be required to understand this relationship further.

Emissions intensity was similar or slightly lower than results from beef production in Europe and North America when results were converted to a LW basis. Emissions intensity (excluding land use and dLUC) from European suckler herds ranged from 11.1 to 15.3 kg CO$2$-e/kg LW (Casey and Holden 2006; Williams et al. 2006; Nguyen et al. 2012). Similarly, emissions from studies of North American beef ranged from 12.7 to 19.2 kg CO$2$-e/kg LW (Beauchemin et al. 2010; Pelletier et al. 2010; Lupo et al. 2013). Where emissions intensity in the present study was lower, this corresponded to lower emissions from energy inputs and nitrous oxide associated with manure or crop production. Nitrous oxide emissions from cropping and manure are lower in Australia than international defaults (DCCEE 2012) because of the low rainfall and high evaporation rates experienced in...
Australia are not favourable to the production of nitrous oxide from crops or pastures.

Conclusions

This study presents data from an assessment of case study and regional datasets, providing a broad, benchmark analysis of resource use and environmental impacts from beef production in two major production regions of Qld and NSW. The study includes energy demand and disaggregated land occupation not previously reported for beef production in Australia. Also included is an expanded analysis of fresh water consumption from case study and regional farms. GHG emissions were found to be similar for different market types and herd productivity explained most of the variability in GHG. Analysis of the case studies suggested higher inputs to improve productivity will result in lower emissions intensity. Fresh water consumption was governed by climate, on-farm supply efficiency and use of irrigation, and the impact of water use was more heavily related to the degree of regional water stress than the volume of water used. Although livestock farmers cannot easily modify local climatic effects that influence water requirements, there may be opportunities to improve supply efficiency on-farm and this may be an effective means to reduce the volume of water used on-farm. Beef cattle were found to rely primarily on non-arable land for grazing, which is unsuitable for most alternative food production systems.

Acknowledgements

This study was funded by Meat and Livestock Australia, and could not have been completed without the contribution of data from the many farmers who provided the case studies presented. Review and input to the project scope from Dr Tom Davison and Dr Beverley Henry was appreciated.

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3.3 ADDENDUM

The results in this paper were modified to enable consistent reporting with other beef and meat production systems by expanding the system boundary to include meat processing, and revising the reference flow/functional unit from ‘kilograms of liveweight’ to ‘kilograms of boneless meat’ (see results in Table 4 and Table 5). Greenhouse gas prediction methods were also standardised to apply the most recent National Inventory Report methods for enteric methane, which slightly reduced the predicted enteric methane from 6.5% to the equivalent of 6.3% of gross energy intake (Charmley et al. 2016, Commonwealth of Australia, 2016).

Meat processing impacts were modelled using inventory data presented in chapter 6 (Table 3) and meat processing assumptions relating to carcase yield were presented in section 2.2.2 of chapter 6.
4 RESOURCE USE AND GREENHOUSE GAS EMISSIONS FROM GRAIN FINISHING BEEF CATTLE IN SEVEN AUSTRALIAN FEEDLOTS: A LIFE CYCLE ASSESSMENT

4.1 ATTRIBUTION STATEMENT

The paper *Resource use and greenhouse gas emissions from grain-finishing beef cattle in seven Australian feedlots: a life cycle assessment* was led by S.G Wiedemann, and co-authored by R. Davis, E.J. McGahan, C.M. Murphy and M Redding. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions
- Development of novel methods for the determination and categorisation of land use
- Integration of primary data into the LCA model, and data analysis
- Preparation of the manuscript and completion of the peer-review process

Davis contributed to the study via:

- Assistance with integrating primary data from previous inventory collection projects into the LCA analysis.
- Review of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data interpretation
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation

Redding contributed to the study and the manuscript via:

- Assistance with integration of manure emission factors into the modelling
- Review of the manuscript
4.2 INTRODUCTION

This paper was the culmination of three major benchmarking studies conducted over 10 years and a major manure GHG emission research program in the feedlot industry. Initial feedlot industry LCA research was used to guide highly detailed water and energy benchmarking research, and also guided research investment to improve knowledge of manure excretion and GHG emissions from feedlots. When published in 2017, the paper was able to draw these previous studies by the author and colleagues into a robust study of the environmental impacts from grain finishing. The paper developed and applied new methods to handle the impact of land use change on water assessment, showing that it was possible to determine water balances for a feedlot-controlled drainage area and compare this to a reference (or “green field”) comparison site.

The study provided robust evidence of lower GHG emissions from grain finished beef compared with grass finished beef using similar market categories and background production systems. However, the study also showed that resource use in the form of energy, water and crop land occupation were all higher for grain finished beef, highlighting the potential trade-offs within this production system.

The study also quantified manure emissions and demonstrated the sensitivity of total emissions to nitrous oxide from the feed pad. Initial findings were used to direct research that later showed this emission source to be less significant than previous inventory values indicated. This finding enabled more robust guidance around mitigating GHG emissions from livestock, and feedlot beef in particular.
Resource use and greenhouse gas emissions from grain-finishing beef cattle in seven Australian feedlots: a life cycle assessment

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Abstract. Grain finishing of cattle has become increasingly common in Australia over the past 30 years. However, interest in the associated environmental impacts and resource use is increasing and requires detailed analysis. In this study we conducted a life cycle assessment (LCA) to investigate impacts of the grain-finishing stage for cattle in seven feedlots in eastern Australia, with a particular focus on the feedlot stage, including the impacts from producing the ration, feedlot operations, transport, and livestock emissions while cattle are in the feedlot (gate-to-gate). The functional unit was 1 kg of liveweight gain (LWG) for the feedlot stage and results are included for the full supply chain (cradle-to-gate), reported per kilogram of liveweight (LW) at the point of slaughter. Three classes of cattle produced for different markets were studied: short-fed domestic market (55–80 days on feed), mid-fed export (108–164 days on feed) and long-fed export (>300 days on feed). In the feedlot stage, mean fresh water consumption was found to vary from 171.9 to 672.6 L/kg LWG and mean stress-weighted water use ranged from 100.9 to 193.2 water stress index eq. L/kg LWG. Mean fossil energy demand ranged from 16.5 to 34.2 MJ lower heating values/kg LWG and arable land occupation from 18.7 to 40.5 m²/kg LWG in the feedlot stage. Mean greenhouse gas (GHG) emissions in the feedlot stage ranged from 4.6 to 9.5 kg CO₂-e/kg LWG (excluding land use and direct land-use change emissions). Emissions were dominated by enteric methane and contributions from the production, transport and milling of feed inputs. Linear regression analysis showed that the feed conversion ratio was able to explain >86% of the variation in GHG intensity and energy demand. The feedlot stage contributed between 26% and 44% of total slaughter weight for the classes of cattle fed, whereas the contribution of this phase to resource use varied from 4% to 96% showing impacts from the finishing phase varied considerably, compared with the breeding and backgrounding. GHG emissions and total land occupation per kilogram of LWG during the grain finishing phase were lower than emissions from breeding and backgrounding, resulting in lower life-time emissions for grain-finished cattle compared with grass finishing.

Additional keywords: agricultural systems, feedlots, global climate change.

Introduction

Grain finishing has become increasingly prevalent in Australia over the past three decades (Bindon and Jones 2001) as a means of increasing cattle growth rates, allowing younger turnoff and potentially heavier finished weights. Market drivers, such as higher carcass weight and marbling in Australia’s international markets have also increased demand for grain-finished cattle. In Australia’s variable climate, feedlots also have an important role to enable finishing of cattle in drought conditions when grass and forage supply is low. However, the impact of grain finishing on finite resources and the environment is a topic of growing interest both in Australia and internationally and this must be considered as part of a broader discussion on the advantages and costs of alternative beef finishing systems.
particularly suited to multi-impact supply chain analysis and has been previously applied to Australian beef case studies (Peters et al. 2010a, 2010b; Eady et al. 2011; Ridoutt et al. 2012; Wiedemann et al. 2016a), as well as wheat (Brock et al. 2012), wool and sheep meat (Brock et al. 2013), pork (Wiedemann et al. 2016b), poultry (Wiedemann and McGahan 2011; Wiedemann et al. 2017) and lamb (Peters et al. 2010a; Wiedemann et al. 2016c) case studies. Wiedemann et al. (2015) showed that water use and GHG emissions associated with beef production have declined in Australia over the past three decades. One major contributing factor to the decline in GHG has been the improved growth rate in young cattle and the increase in slaughter weight across the national herd (Wiedemann et al. 2015). These increases in productivity result in relatively shorter feeding periods to achieve desirable finishing weights, resulting in a decrease in total emissions over the lifetime of an animal. The most significant change in the industry contributing to these effects has been the large increase in grain finished cattle over this time period. Although several studies have compared grain and grass finishing (Peters et al. 2010a; Pelletier et al. 2010), few studies have provided a detailed analysis of the gate-to-gate grain finishing system specifically.

Key differences between grass finishing and grain finishing in feedlots include lower GHG emissions from enteric methane (Dong et al. 2006) and potentially higher emissions from manure management (Commonwealth of Australia 2015b) for feedlot-finished cattle. Manufacture of feed inputs requires inputs of fertiliser and machinery operations, increasing fossil fuel energy use and potentially GHG emissions compared with grass finishing. Transport-related effects are also evident. Feedlot operations and feed preparation are also energy intensive processes associated with grain finishing. Fossil fuel and water inputs for grain milling and feedlot operations have been quantified for Australian feedlots by Davis et al. (2010a, 2010b). However, these studies did not include some aspects of the wider feedlot system, such as the production of feed inputs and therefore a comprehensive understanding of water use for grain finishing has not been completed to date. The present study aimed to (i) determine resource use and GHG emissions from Australian beef finished in feedlots, targeting three different market specifications, (ii) to provide benchmarking data and identify the relative impacts of different inputs and processes on GHG and resource use, and (iii) to quantify the impact of different finishing systems on the full cradle-to-gate impacts for three different market types commonly fed in Australia.

Materials and methods

Goal and scope

This study reports a gate-to-gate LCA of seven beef feedlots in eastern Australia. It is a companion study to Wiedemann et al. (2016a), which used LCA to investigate environmental impacts from grass-finished beef production in eastern Australia to the farm gate. The study investigated global warming (aggregated GHG emissions, including impacts from land use and direct land use change – LU and dLUC). Fossil fuel energy demand was assessed using the FossilFuelDepletion indicator (Frischknecht et al. 2007), measured in mega-joules (MJ) using lower heating values (LHV). Stress-weighted water use was assessed using the water stress index (WSI) values from Pfister et al. (2009). Results from two life cycle inventory (LCI) methods were also assessed: fresh water consumption and land occupation. Results were presented using a functional unit (FU) of kilograms of liveweight gain (kg LWG) in the grain finishing phase (gate-to-gate) and impacts from cradle-to-gate were assessed using data for feeder cattle reported by Wiedemann et al. (2016a) and reported per kilogram of liveweight (LW) immediately before meat processing. The gate-to-gate analysis included all processes and inputs associated with grain finishing in commercial feedlots, including the impacts from producing the ration, feedlot operations, transport, and livestock emissions while cattle are in the feedlot.

Description of the case-study feedlots

The seven feedlots (FL) ranged in throughput from 6000 to 130 000 head of cattle annually, with a total throughput for the years 2007 and 2008 of 330,000 comprising ~15% of industry throughput for these years (ALFA and MLA 2007, 2008). Feedlots were located in: central Queensland (Qld – FL 1), southern Qld (FL 2, 3, 4), northern New South Wales (NSW – FL 5), southern NSW (FL 6) and Victoria (Vic. – FL 7). Cattle fed to three different market specifications were investigated: short-fed domestic market (55–80 days on feed), mid-fed export (108–164 days on feed) and long-fed export (>300 days on feed).

Description of supply chains

Average feeder cattle production systems from northern NSW were used to provide impacts associated with breeding and backgrounding cattle before feedlot entry. Data were modelled from the NSW regional average farm dataset described in Wiedemann et al. (2016a), and differed only with respect to the feedlot entry weight requirements for the different market classes. Although breed and genotype differences may exist between cattle entering different grain-finished markets, data were not available to differentiate between these herds at the farm and backgrounding stages, and differences are not expected to be large. Impacts associated with LU and dLUC in the breeding and backgrounding stage were determined using data from Henry et al. (2015).

Life cycle inventory

Detailed production data, livestock inventories and input data were collected from feedlot financial records, interviews and site visits (Davis et al. 2010a, 2010b). Livestock data were reported from feedlot records of the number and weight of cattle entering and leaving the feedlot, and the recorded LWG and dry matter intake (DMI). LWG was corrected for shrink in cattle entering the feedlot to avoid over estimating LWG during the feeding period. Production characteristics are provided in Tables 1 and 2. Fossil fuel energy demand was determined from inventories of purchased fossil fuels and electricity use at the feedlot (Tables 3, 4) and from impacts associated with commodity use. Major energy inputs at the feedlot were mainly related to feed processing (electricity and gas use) and diesel for vehicle operation, including pen cleaning and feed delivery. Transport records for livestock movements,
purchased inputs and staff movements were determined from feedlot inventory data and were included in the analysis. Impacts associated with infrastructure, such as feedlot construction, were excluded based on the findings of a scoping study showing the contribution from these was <1% (S. G. Wiedemann, R. J. Davis and E. J. McGahan, unpubl. data). Impacts associated with services such as communications, insurance and accounting, were modelled based on expenditure, using economic input-output data (Rebitzer et al. 2002).

Dry matter intake was determined from records of feed delivery to the feed bunk at each feedlot, converted from an as-fed basis to DMI using recorded or calculated ration moisture contents. Feed intake and feed characteristics were used to model livestock GHG emissions, using methods described below. Background data were sourced from the Australian LCI database (Life Cycle Strategies 2007) where available, or the European Ecoinvent (2.2) database (Swiss Centre for Life Cycle Inventories 2010). Feed grain inventory processes were detailed in Wiedemann et al. (2016b) and Wiedemann and McGahan (2011). Commodity data, and inputs associated with feed milling, are provided in Tables 4 and 5, respectively.

**Fresh water consumption**
The water-use inventory was developed using measured water data described in Davis et al. (2010a); and modelling of dam water supply systems and dam supply efficiencies using

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**Table 1. Production parameters for three market types of feedlot-finished cattle**

<table>
<thead>
<tr>
<th></th>
<th>Domestic grain short-fed (n = 5)</th>
<th>Export grain mid-fed (n = 4)</th>
<th>Export grain long-fed (n = 1)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Mean</td>
<td>Range</td>
<td>Mean</td>
</tr>
<tr>
<td>Entry weight (kg LW)</td>
<td>347</td>
<td>327–370</td>
<td>421</td>
</tr>
<tr>
<td>Average daily gain (kg/day)</td>
<td>1.75</td>
<td>1.7–1.9</td>
<td>1.84</td>
</tr>
<tr>
<td>Feed conversion ratio (kg/kg – dry matter basis)</td>
<td>5.2</td>
<td>4.6–5.7</td>
<td>5.8</td>
</tr>
<tr>
<td>Mortality rate</td>
<td>0.8%</td>
<td>0.16–1.65%</td>
<td>0.7%</td>
</tr>
<tr>
<td>Dry matter intake (kg/head.day)</td>
<td>9.0</td>
<td>8.5–9.5</td>
<td>10.7</td>
</tr>
</tbody>
</table>

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**Table 2. Production parameters for breeding, backgrounding and finishing of grain fed cattle**

<table>
<thead>
<tr>
<th></th>
<th>Short-fed domestic feedlot finished</th>
<th>Mid-fed export feedlot finished</th>
<th>Long-fed export feedlot finished</th>
</tr>
</thead>
<tbody>
<tr>
<td>Days on farm</td>
<td>365</td>
<td>365</td>
<td>365</td>
</tr>
<tr>
<td>Average daily gain (kg)</td>
<td>0.63</td>
<td>0.63</td>
<td>0.63</td>
</tr>
<tr>
<td>Closing liveweight (kg – young cattle)</td>
<td>263</td>
<td>263</td>
<td>263</td>
</tr>
<tr>
<td>Closing liveweight (kg – breeding cattle)</td>
<td>125</td>
<td>125</td>
<td>125</td>
</tr>
<tr>
<td>Days on farm</td>
<td>153</td>
<td>338</td>
<td>342</td>
</tr>
<tr>
<td>Average daily gain</td>
<td>0.55</td>
<td>0.47</td>
<td>0.52</td>
</tr>
<tr>
<td>Closing liveweight (kg – backgrounding cattle)</td>
<td>347</td>
<td>421</td>
<td>441</td>
</tr>
<tr>
<td>Days on feed – feedlot</td>
<td>69</td>
<td>125</td>
<td>346</td>
</tr>
<tr>
<td>Average daily gain</td>
<td>1.75</td>
<td>1.84</td>
<td>0.99</td>
</tr>
<tr>
<td>Closing liveweight (kg – finishing cattle)</td>
<td>468</td>
<td>652</td>
<td>784</td>
</tr>
</tbody>
</table>

A Data sourced from NSW regional average farm (RAF) dataset reported in Wiedemann et al. (2016a).

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**Table 3. Feedlot purchased energy for general operations (excludes feed-milling), reported per 1000-head days**

<table>
<thead>
<tr>
<th></th>
<th>FL1</th>
<th>FL2</th>
<th>FL3</th>
<th>FL4</th>
<th>FL5</th>
<th>FL6</th>
<th>FL7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity (kWh/1000-head dayA)</td>
<td>19.55</td>
<td>58.37</td>
<td>63.84</td>
<td>84.73</td>
<td>17.01</td>
<td>19.84</td>
<td>68.83</td>
</tr>
<tr>
<td>Diesel (L/1000-head day)</td>
<td>11.34</td>
<td>16.75</td>
<td>12.14</td>
<td>19.50</td>
<td>7.66</td>
<td>13.86</td>
<td>12.67</td>
</tr>
<tr>
<td>Gas (MJ/1000-head day)</td>
<td>0.83</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
<td>0.00</td>
</tr>
<tr>
<td>Petrol (L/1000-head day)</td>
<td>0.72</td>
<td>0.81</td>
<td>8.16</td>
<td>9.65</td>
<td>11.03</td>
<td>11.76</td>
<td>6.90</td>
</tr>
</tbody>
</table>

A Head days represent one animal occupying the feedlot for 1 day.
methods outlined in Wiedemann et al. (2016a). Details of the water supply system and system efficiency are provided in Table 6. Water supply was predominantly from bores located on-site, though dams were also an important water source. All dams either captured overland flow from the local area, or were pumped from local river systems.

Australian feedlots are designed to control drainage and overland flow around the feedlot site to restrict movement of manure nutrients to the environment (Skerman 2000), thus restricting runoff to the environment compared with the reference site before feedlot construction. The change in runoff associated with LUC was included in the fresh water consumption losses associated with on-site irrigation were also determined from farm water balances following methods described in Wiedemann et al. (2016a). Methods described by Wiedemann et al. (2015) were used to model water use associated with irrigation used in the production of purchased feed. Irrigation water used to produce hay and silage at each feedlot was recorded as part of the survey. Irrigation water use associated with water-intensive inputs such as cotton seed, a co-product of cotton lint production, was modelled using inventory data from Tan et al. (2013) and water-use data from ABS (2011). Water-use was allocated between cotton lint and seed on an economic basis, with 10% of total irrigation water use attributed to cotton seed production.

### Stress-weighted water use

Stress-weighted water use is a weighted indicator of the volume of fresh water used by the system in each region with the appropriate WSI values from Pfister et al. (2009). The value was then divided by the global average WSI (0.602) and expressed as a water equivalent (H₂O-e; Ridoutt and Pfister 2010).

### Land occupation

Land occupation was determined from records regarding the size of each feedlot and the land occupation associated with feed production. The land footprint of each feedlot (Table 6) was determined using geographic information systems (GIS) software and satellite imagery, whereas land occupation data for feed inputs were derived from inventory datasets. No characterisation factors were applied, and land occupation results were reported in square metre years (m² year).

### Livestock and manure GHG emissions

Enteric methane was modelled using the regression of Moe and Tyrrell (1979) as applied by the Australian National Inventory Report (NIR) (Commonwealth of Australia 2015b). To examine the sensitivity of total feedlot emissions to this factor, a sensitivity analysis was performed by re-calculating enteric methane using IPCC 2006 factors of 3% gross energy intake (GEI) and 6.5% GEI (Dong et al. 2006). Manure emission methods were determined using methods outlined in the NIR (Commonwealth of Australia 2015b) and are shown in Table 8. Uncertainty estimates for manure emission factors were based on the IPCC (Dong et al. 2006). Global warming potentials of 25 for methane and 298 for nitrous oxide (Solomon et al. 2007) were applied.

### Table 4. Feed-milling energy and water inputs per tonne of ration milled and delivered to the feed bunk

<table>
<thead>
<tr>
<th>Ration component</th>
<th>FL1</th>
<th>FL2</th>
<th>FL3</th>
<th>FL4</th>
<th>FL5</th>
<th>FL6</th>
<th>FL7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity (kwh/tonne)</td>
<td>8.05</td>
<td>6.61</td>
<td>4.70</td>
<td>7.08</td>
<td>6.68</td>
<td>5.54</td>
<td>5.36</td>
</tr>
<tr>
<td>Diesel (L/tonne)</td>
<td>0.85</td>
<td>0.83</td>
<td>2.63</td>
<td>2.27</td>
<td>0.62</td>
<td>1.77</td>
<td>1.55</td>
</tr>
<tr>
<td>Gas (MJ/tonne)</td>
<td>180.65</td>
<td>128.90</td>
<td>0.00</td>
<td>0.00</td>
<td>163.91</td>
<td>163.21</td>
<td>191.59</td>
</tr>
<tr>
<td>Petrol (L/tonne)</td>
<td>138.89</td>
<td>130.61</td>
<td>189.36</td>
<td>42.44</td>
<td>194.29</td>
<td>80.46</td>
<td>197.27</td>
</tr>
</tbody>
</table>

- AProtein content in brackets.
- BIncludes lucerne, sorghum and wheat hay.
- CIncludes corn, wheat, triticale and sorghum straw.
- DIncludes corn, sorghum, oats and wheat silage.

### Table 5. Commodity purchases per tonne of ration (as-fed) for three market classes, averaged over 2 years

<table>
<thead>
<tr>
<th>Ration component</th>
<th>Short-fed</th>
<th>Mid-fed</th>
<th>Long-fed</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ration component</td>
<td>domestic</td>
<td>export</td>
<td>export</td>
</tr>
<tr>
<td>Barley (10%)</td>
<td>75.7</td>
<td>94.7</td>
<td>136.7</td>
</tr>
<tr>
<td>Maize (8%)</td>
<td>68.7</td>
<td>94.7</td>
<td>136.7</td>
</tr>
<tr>
<td>Sorghum (10%)</td>
<td>311.2</td>
<td>289.4</td>
<td>157.3</td>
</tr>
<tr>
<td>Wheat (13%)</td>
<td>225.6</td>
<td>295.8</td>
<td>309.3</td>
</tr>
<tr>
<td>Canola meal (36%)</td>
<td>0.7</td>
<td>9.0</td>
<td>1.3</td>
</tr>
<tr>
<td>Cottonseed meal (38%)</td>
<td>20.1</td>
<td>23.9</td>
<td>0.0</td>
</tr>
<tr>
<td>White flufly cottonseed</td>
<td>61.8</td>
<td>63.4</td>
<td>0.0</td>
</tr>
<tr>
<td>Hay</td>
<td>26.1</td>
<td>33.6</td>
<td>2.9</td>
</tr>
<tr>
<td>Straw</td>
<td>10.6</td>
<td>3.2</td>
<td>89.0</td>
</tr>
<tr>
<td>Silage</td>
<td>111.7</td>
<td>85.1</td>
<td>199.0</td>
</tr>
<tr>
<td>Cotton hulls</td>
<td>4.3</td>
<td>17.2</td>
<td>0.0</td>
</tr>
<tr>
<td>Oil</td>
<td>7.2</td>
<td>12.2</td>
<td>1.6</td>
</tr>
<tr>
<td>Molasses</td>
<td>25.1</td>
<td>21.9</td>
<td>71.2</td>
</tr>
<tr>
<td>Dry supplement</td>
<td>0.0</td>
<td>0.0</td>
<td>31.7</td>
</tr>
<tr>
<td>Wet supplement</td>
<td>51.2</td>
<td>58.7</td>
<td>0.0</td>
</tr>
<tr>
<td>Total (kg)</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
</tr>
</tbody>
</table>

- AReported on an ‘as-fed’ basis inclusive of moisture.
As noted by Redding et al. (2015), the NIR method for feed pad nitrous oxide is not substantiated by adequate field research. Redding et al. (2015) found feed pad nitrous oxide emissions differed in both the mechanisms influencing emissions and the intensity of emissions compared with the Australian inventory approach. To examine the sensitivity of total feedlot emissions to this factor, we applied a comparison method to predict nitrous oxide from manure deposited to the feed pad, using data from Redding et al. (2015) and P. Shorten and M. Redding (unpubl. data). This method used climate-specific nitrous oxide emission factors, based on emissions per m² of pen surface area occupied per animal (stocking density) in each feedlot region (see Table 8).

**GHG emissions (including dLUC) – feedlot operations and feed inputs**

The GHG emissions associated with fossil fuel use at the feedlot were determined from the inventory of fossil fuel use in Davis et al. (2010b). GHG emissions arising from grain or forage production were determined from the databases described. Previous grain studies reported in Wiedemann et al. (2010) and Brock et al. (2012) have not included soil carbon losses associated with LU and dLUC and emissions from these sources remain uncertain, though inclusion of these sources is recommended for feed grains (LEAP 2014). Two methods were used to determine potential emissions from these sources. First, we determined total LU and dLUC, emissions from an analysis of data from the Australian national inventory (Commonwealth of Australia 2015c). This showed annualised, national emissions associated with conversion of forest land to crop land (dLUC) were –4 755 000 t CO₂-e in the period 1990–2010 (Commonwealth of Australia 2015a). National LU emissions from crop land were –4 800 000 t CO₂-e (negative emissions indicate carbon sequestration), annualised over the same period. Carbon sequestration in Australian crop land is mostly in response to carbon sequestration resulting from adoption of improved cropping practices over the past 20 years, as shown by Luo et al. (2010). When divided by the average total land area sown to crops annually in Australia over the period 1990–2010, annualised emissions from LU and dLUC were –229 and 227 kg CO₂-e/ha. Alternatively, LU and dLUC emissions were estimated using a method adapted from Wiedemann et al. (2016c). This accounted for different levels of carbon loss for crop land managed with tillage or zero tillage in each state, and accounted for dLUC as a result of the expansion of crop land in the eastern states of Australia (Lesslie and Mewett 2013; Wiedemann et al. 2016c). Total reported LU and dLUC emissions applied to crop land using this method were 0.44, 0.51 and 0.67 t CO₂-e/ha.year for NSW, southern NSW/Vic., and Qld, respectively, and a detailed explanation of GHG emissions from LU and dLUC is presented in the supplementary material.

**Handling co-production**

The feedlot system produced beef and manure. Manure is sold as a fertiliser replacement and soil conditioner and has a very low value compared with beef. Co-production of manure was handled by system expansion taking into account the avoided fertiliser for crop production that would be required in the absence of feedlot manure, using the method described by Wiedemann et al. (2010). This method accounted for the nitrogen and phosphorus content in manure, and the relative emissions from manure or avoided fertiliser use, after losses associated with spreading were taken into account.

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### Table 6. Land and water resources for seven feedlots in eastern Australia

<table>
<thead>
<tr>
<th>Location</th>
<th>FL1</th>
<th>FL2</th>
<th>FL3</th>
<th>FL4</th>
<th>FL5</th>
<th>FL6</th>
<th>FL7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Location</td>
<td>Qld</td>
<td>Qld</td>
<td>Qld</td>
<td>Qld</td>
<td>NSW</td>
<td>NSW</td>
<td>Vic.</td>
</tr>
<tr>
<td>Average annual rainfall (mm)</td>
<td>555</td>
<td>582</td>
<td>624</td>
<td>662</td>
<td>526</td>
<td>857</td>
<td>430</td>
</tr>
<tr>
<td>Land use</td>
<td>Feedlot area – non-arable (ha)</td>
<td>102.0</td>
<td>47.7</td>
<td>76.0</td>
<td>25.9</td>
<td>71.0</td>
<td>149.0</td>
</tr>
</tbody>
</table>

### Table 7. Runoff from reference land occupation attributed to feedlot cattle production

<table>
<thead>
<tr>
<th>Runoff from reference land occupation (ML/year)</th>
<th>FL1</th>
<th>FL2</th>
<th>FL3</th>
<th>FL4</th>
<th>FL5</th>
<th>FL6</th>
<th>FL7</th>
</tr>
</thead>
<tbody>
<tr>
<td>Runoff from reference land occupation (ML/year)</td>
<td>49.0</td>
<td>14.5</td>
<td>24.3</td>
<td>2.8</td>
<td>14.1</td>
<td>23.1</td>
<td>10.4</td>
</tr>
<tr>
<td>Fresh water consumption attributed to cattle production (L/head finished)</td>
<td>1069.8</td>
<td>119.9</td>
<td>719.7</td>
<td>389.1</td>
<td>283.5</td>
<td>877.6</td>
<td>232.0</td>
</tr>
</tbody>
</table>

ARecorded for nearest major town.
The study was modelled using SimaPro 8 (Pré-Consultants 2014). Model uncertainty for the manure management emissions was assessed using Monte Carlo analysis in SimaPro 8. One-thousand iterations provided a 95% confidence interval for the results. Comparison between the short-fed and mid-fed market types was made using the t-test in R (R Development Core Team 2014). Results from the single long-fed market type analysis are described qualitatively. Linear regression analysis was used to describe the influence of key production parameters and farm variables on GHG emissions and resources using R.

Results

Fresh water consumption and stress-weighted water use – feedlot gate-to-gate

Mean fresh water consumption was dominated by irrigation water use and supply losses. Off-site irrigation and supply losses contributed 35–57% of total fresh water consumption (see Fig. 1), whereas on-site irrigation and supply losses contributed 43–56% (Fig. 1). No correlation was found between fresh water use and feed conversion ratio or average daily gain. Mean fresh water consumption was not significantly different between the short-fed and mid-fed cattle (see Table 9). Fresh water consumption was lower from the long fed feedlot because of lower on-site irrigation and the lower quantities of specific, water intensive ration commodities used.

Land occupation – feedlot gate-to-gate

Mean crop land occupation was highest for the long-fed cattle (40.5 m²/kg LWG), whereas the land occupation for the

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Table 8. Greenhouse gas emission factors with uncertainty for modelling livestock emissions from feedlot cattle

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Emission and units</th>
<th>Key parameters/model</th>
<th>Assumed uncertainty</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enteric fermentation</td>
<td>kg CH₄/head</td>
<td>3.406 + 0.510SR + 1.736H + 2.648C</td>
<td>SR: 0.23–0.79</td>
<td>Commonwealth of Australia (2015b); Moe and Tyrrell (1979)</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>H: 1.47–2.00</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>C: 2.46–2.83</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from pen surface</td>
<td>MCF</td>
<td>3% (Qld) ±20%</td>
<td></td>
<td>Commonwealth of Australia (2015)</td>
</tr>
<tr>
<td></td>
<td>NH₃-N/kg N excreted</td>
<td>0.60 ±50%</td>
<td></td>
<td>Commonwealth of Australia (2015); Dong et al. (2006)</td>
</tr>
<tr>
<td></td>
<td>N₂O-N/kg N excreted</td>
<td>0.02 Factor of 2</td>
<td></td>
<td>Commonwealth of Australia (2015); Dong et al. (2006)</td>
</tr>
<tr>
<td>Manure – emissions from stockpile</td>
<td>MCF</td>
<td>2% ±20%</td>
<td></td>
<td>Commonwealth of Australia (2015b); Watts et al. (2012)</td>
</tr>
<tr>
<td></td>
<td>NH₃-N/kg N to stockpile</td>
<td>0.25 ±25%</td>
<td></td>
<td>Commonwealth of Australia (2015b); Dong et al. (2006)</td>
</tr>
<tr>
<td></td>
<td>N₂O-N/kg N to stockpile</td>
<td>0.005 Factor of 2</td>
<td></td>
<td>Commonwealth of Australia (2015b); Dong et al. (2006)</td>
</tr>
<tr>
<td>Effluent – from anaerobic storage</td>
<td>MCF</td>
<td>77% (Qld) ±20%</td>
<td></td>
<td>Commonwealth of Australia (2015b); Dong et al. (2006)</td>
</tr>
<tr>
<td></td>
<td>NH₃-N/kg N to pond</td>
<td>0.35 0.25–0.75</td>
<td></td>
<td>Commonwealth of Australia (2015b); Dong et al. (2006)</td>
</tr>
<tr>
<td>Land application</td>
<td>N₂O-N/kg N to land</td>
<td>0.01 0.003–0.03</td>
<td></td>
<td>De Klein et al. (2006); Commonwealth of Australia (2015b)</td>
</tr>
<tr>
<td>Indirect N₂O from volatilised NH₃</td>
<td>N₂O-N/kg N volatilised as NH₃</td>
<td>0.2 0.18–0.22</td>
<td></td>
<td>Watts et al. (2012)</td>
</tr>
<tr>
<td>Sensitivity analyses</td>
<td>Enteric methane</td>
<td>CH₄% of Gross energy intake</td>
<td>±1%</td>
<td>Dong et al. (2006)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>3% – high concentrate diet (domestic and mid-fed), 6.5% low concentrate diet (long fed)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Manure – from pen surface</td>
<td>N₂O/m² day</td>
<td>Factor of 2</td>
<td>Redding et al. (2015); P. Shorten and M. Redding (unpubl. data)</td>
</tr>
<tr>
<td></td>
<td>0.039 (Nth Qld)</td>
<td>0.033 (Sth Qld)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.03 (Nth Qld)</td>
<td>0.012 (Sth NSW)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.016 (Sth NSW)</td>
<td>0.005 (Vic.)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Stocking density</td>
<td>m²/head</td>
<td></td>
<td>Feedlot data</td>
</tr>
<tr>
<td></td>
<td>16.3 (FL1, FL5)</td>
<td>12.2 (FL2)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>24.5 (FL3)</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>12.8 (FL4)</td>
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<td></td>
<td></td>
<td>18.2 (FL6)</td>
<td></td>
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<tr>
<td></td>
<td></td>
<td>17.5 (FL7)</td>
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</tr>
</tbody>
</table>
short-fed (18.7 m²/kg LWG) and mid-fed cattle (22.5 m²/kg LWG) was considerably lower (see Table 9 and Fig. 2).

Energy demand – feedlot gate-to-gate

Mean fossil fuel energy demand was significantly lower ($P < 0.05$) for the short-fed cattles (16.5 MJ LHV/kg LWG) than the mid-fed cattle (18.9 MJ LHV/kg LWG). Energy demand was considerably higher in the long-fed market class (see Table 9 and Fig. 3). Feed production and milling were the largest contributors to energy demand. Among individual feedlots, energy demand varied by 43% between the lowest and highest values for the short-fed and mid-fed market classes. The linear regression of energy demand and feed conversion ratio showed that this parameter explained 0.86 of the variability ($P < 0.001$) according to the following equation:

$$
\text{Energy demand} = 3.34 \pm 0.36 \text{ feed conversion ratio} - 0.83 \pm 1.69 \left( R^2 = 0.86 \right)
$$

Although the small sample size ($n = 1$) precluded statistical comparison of the long-fed system with the shorter feeding periods, there was a strong trend towards lower energy impacts from the shorter feeding periods.

GHG emissions – feedlot gate-to-gate

The mean estimated GHG intensity (excluding LU and dLUC) was 4.6 and 5.2 kg CO$_2$-e/kg LWG for the short-fed and mid-fed cattle with a 30% variation between the highest and lowest values between feedlots. Emissions from short-fed and mid-fed cattle were lower than for long-fed cattle, and were significantly different from each other ($P < 0.05$). The GHG contribution analysis showed mean contribution (excluding land use change) by gas type was similar across market classes. When analysed by gas type, methane contributed 51% predominantly from enteric sources, CO$_2$ from fossil fuels contributed 22%, and nitrous oxide from manure management and cropping contributed 27% of total impacts (Fig. 4). When emissions were grouped according to different subsystems, 26% of GHG impacts arose from ration production, transport and milling, 3% from feedlot operations (excluding milling), 23% from manure emission sources, and 48% from livestock enteric methane.
The linear regression analysis showed that feed conversion ratio was able to explain 0.95 of the variation ($P < 0.001$) in GHG intensity. The regression model of GHG emissions intensity was:

$$\text{GHG} = 0.979 \pm 0.06 \text{ feed conversion ratio} - 0.50 \pm 0.36 \ (R^2 = 0.95)$$

Estimated emissions intensity was sensitive to the method chosen for predicting livestock emissions. Applying IPCC defaults for predicting enteric methane revealed ~40% lower enteric methane for the domestic and mid-fed cattle per kg LWG, whereas enteric methane from long-fed cattle increased by 26% with the IPCC default method applied. Results were sensitive to the factors used to predict manure GHG emissions. Applying alternative emission factors for pen nitrous oxide resulted in a decrease in estimated feedpad nitrous oxide from 75.0 to 14.0 kg CO$_2$-e/100 kg LWG for the southern Qld feedlot, and from 101.1 to 5.3 kg CO$_2$-e/100 kg LWG for the southern NSW feedlot. This resulted in total manure management emissions decreasing 60% and 81% for the southern Qld and southern NSW feedlots, respectively. The impact on total emissions from all sources (excluding LU and dLUC), when the sensitivity analysis emission factors from Table 8 were used, showed a decrease of 16% for the short-fed market class, 18% for the mid-fed, and a 19% decrease for the long-fed market class with this revised factor applied.

Manure management emissions also have a high degree of uncertainty. Estimated manure emissions and mean results calculated using a Monte Carlo analysis of uncertainty are shown in Table 10. Calculated totals were slightly different than the mean results from the Monte Carlo analysis, because the uncertainty analysis applied skewed distributions for some factors such as nitrous oxide from land application of manure, and emissions of indirect nitrous oxide from ammonia volatilisation.
Mean emissions from LU and dLUC sources differed depending on the method applied. Where national estimates of LU and dLUC were applied, emissions from this source were negligible. Where state-based estimates of dLUC and high estimates of carbon loss from LU were applied, emissions were 1.19, 1.32 and 1.8 kg CO2-e/kg LWG for the short-fed, mid-fed and long-fed cattle respectively, with differences associated with feed conversion ratio and grain production region.

Resources and GHG – cradle-to-farm gate

Liveweight gain during the feedlot finishing phase represents 26%, 35% and 44% of the total beef production from the primary production system, for the short-fed domestic, mid-fed export and long-fed export respectively. However, the feedlot finishing phase contributed 34–72% of total energy demand, 80–96% of total crop land occupation, and 38–53% of fresh water consumption (see Fig. 5). Total energy demand was 10.0, 11.6 and 18.1 MJ per kilogram of LW finished for the short-fed domestic, mid-fed export and long-fed export respectively. Fresh water consumption was 296, 308 and 206 L, water stress was 99, 118 and 18 L H2O-e, total land occupation was 106, 97 and 101 m²/kg LW, and crop land occupation was 5.6, 8.9 and 24 m²/kg LW finished for the short-fed domestic, mid-fed export and long-fed export, respectively. The feedlot finishing phase was found to contribute a relatively smaller proportion of overall impacts for total land occupation, at just 4%, 8% and 23% for the short-fed domestic, mid-fed export and long-fed export, respectively.

Greenhouse gas emissions were found to contribute a relatively smaller proportion of overall impacts (Fig. 5). Excluding LU and dLUC, GHG emissions were higher (9.9, 9.4 and 10.6 kg CO2-e/kg LW finished for the short-fed domestic, mid-fed export and long-fed export, respectively), compared with the feedlot stage only. With LU and dLUC emissions included, results varied depending on the assumptions used. Where modest levels of vegetation carbon sequestration were included in the breeding and backgrounding stage (after Henry et al. 2015), LU and dLUC emissions were negligible for all market types. In this scenario, sequestration in the breeding and backgrounding stages compensated for emissions from crop land associated with feedlot ration production. Where some emissions from vegetative management were included in the breeding and backgrounding stages (after Henry et al. 2015), LU and dLUC emissions were 0.2–0.3 kg CO2-e or 1.5–2.0 kg CO2-e/kg LW finished depending on the method applied for estimating crop land emissions.

Discussion

The number of cattle finished on grain in Australia has increased significantly over the past 30 years. The net impact of this change has been a reduction in GHG emissions across the Australian herd (Wiedemann et al. 2015), in response to a shift from higher emission intensity grass-finished cattle, to lower emission intensity grain-finished cattle. However, increased production intensity via lot feeding has contributed to higher energy use (Wiedemann et al. 2015) and the impact on water use has not been studied in detail. This study is the first to provide a detailed analysis of resource use and GHG emissions of the grain finishing component of the supply chain for different market types. In the grain finishing stage, we found the feed conversion ratio, a major productivity factor for the industry, explained a high proportion of the variability in GHG emissions intensity, energy demand and arable land occupation.
Fig. 5. Contribution of different stages of the supply chain to greenhouse gas emissions and resource use for grain-finished cattle.
but not fresh water consumption or stress weighted water use. A significant but lesser association was also found between average daily gain and resource use and emissions intensity (results not shown). Fresh water consumption and stress-weighted water use were more heavily influenced by the use of irrigation for particular feed types such as cotton seed and regional water stress, rather than differences in livestock productivity.

Resource use – feedlot gate-to-gate

Water is a critical input for livestock production. A detailed analysis of water use at Australian feedlots by Davis et al. (2010a) showed that water use within the feedlot complex is dominated by cattle drinking water, with smaller contributions from feed management, cattle washing and other minor uses. In the present study, we used this dataset but expanded the analysis to include supply losses and irrigation water use both on-site and off-site. The fresh water consumption results reported by Davis et al. (2010a) converted to a LWG basis ranged from 22.5 to 32 L/kg LWG, which represented only ~10% of the fresh water consumption when irrigation and supply losses were included in the analysis (Fig. 1).

We found irrigation (including supply losses) to be the largest source of fresh water consumption, with mean contributions across the market types of between 57% and 91% for the feedlot finishing stage. Specifically, large volumes of water were associated with the production of cotton seed and cotton seed meal, which was a common feed input (~6% of ration) for the short-fed and mid-fed market types (see Table 5) and alone accounted for 25–35% of total fresh water consumption for these market types. In addition, the irrigation of hay and silage on feedlot farms consumed large volumes of water. This study found mean on-site irrigation (including supply losses) to be 49% of total fresh water consumption across all feedlots and market types. Among individual feedlots the contribution of irrigation water ranged from zero to 898.8 L/kg LWG in response to local irrigation water availability, which varied from feedlot to feedlot and was not strongly related to market type. Because of the wide range in irrigation water use between feedlots, no association was found between fresh water consumption and productivity for each market type. As only one feedlot was used to assess the long-fed market type, the total quantum, and contribution to water use may not be representative of this market type and these results should be viewed with caution.

Fossil fuel energy demand was dominated by feed production and milling rather than feedlot operations. When analysed separately, feed milling contributed 12% to overall energy demand whereas other feedlot operations contributed 11%. Although milling and general feedlot operations are the only parts of the operation under the direct control of feedlot operators, the total energy demand associated with production is more heavily influenced by feed conversion ratio because of the indirect influence of energy associated with feed production and transport. Total energy demand was considerably higher than reported by Davis et al. (2010b), who reported direct energy demand of 2.1–5.7 MJ for feedlot operations, but excluded energy demand associated with feed production.

Land occupation was dominated by feed production, whereas the occupation for the feedlot itself was insignificant because of the very high density of livestock on relatively small land areas.

GHG emissions – feedlot gate-to-gate

The GHG from sources other than LU and dLUC were dominated by enteric methane and emissions from manure, with smaller contributions from nitrous oxide from cropping and fossil fuel-related CO₂ (Fig. 4). Emissions intensity was largely governed by productivity factors, which influenced enteric methane, manure and feed-related emissions. The significance of ADG on emissions intensity was shown by Hunter and Nieth (2009) and Wiedemann et al. (2016a) and was confirmed in the present study (results not shown), though stronger regression relationships were found to feed conversion ratio here. Impacts from LU and dLUC were substantially different depending on the method applied, resulting in negligible change, or a substantial increase in GHG if LU soil carbon losses were assumed to be high. When the high estimates from these emission sources were included in the contribution analysis, impacts associated with feed production, transport and milling increased to between 38% and 42% of emissions in the gate-to-gate analysis. Considering the potential significance of LU and dLUC emissions, further research is required to improve this analysis for feed grain inventories in Australia.

Sensitivity to emission factors for GHG prediction

Emission intensity prediction is sensitive to specific emission estimation methods. The method used to calculate enteric methane emissions in the Australian GHG inventory predicts significantly higher emissions than the international default IPCC method (Dong et al. 2006). However, the Australian method has been found to predict emissions that are in closer agreement with the measurements made by McGinn et al. (2008) at one Qld feedlot, and are thus considered a robust representation of Australian feedlot production. We note that the Australian method predicts emissions (relative to GEI – 4.9–5.3%) that are mid-way between the recommended IPCC factors of 3% GEI for ~90% concentrate diets and 6.5% for diets with less than ~90% concentrates. For the domestic and mid-fed market types, the Australian method is therefore more conservative than the IPCC approach, though this is not the case for the long-fed market type. Further research would therefore be warranted to ascertain the level of agreement between predicted emissions using the preferred Australian inventory method, and measured enteric methane from long-fed cattle with lower concentrate diets. We found predicted nitrous oxide emissions from manure deposited to the feed pad to be a sensitive model parameter. Recent Australian research (Redding et al. 2015) demonstrated that emissions may be considerably lower from this source than those predicted using the NIR factors, and could result in total emissions per kilogram of beef being 16–19% lower than reported here. Further Australian research is required to confirm nitrous oxide from this source.

Cradle to farm gate

Feedlot finishing of beef cattle produces a substantial productivity increase compared with the grass-fed stages.
before the feedlot. Daily growth rates in the present study were ~3 times higher in the finishing phase compared with backgrounding or finishing on pasture, and finished weights are consequently achieved at a younger age than comparative grass-finished cattle. Compared with the companion analysis of grass-finished beef by Wiedemann et al. (2016a), GHG emissions excluding LU and dLUC were 15%, 23% and 13% lower for the domestic, mid-fed and long-fed export cattle compared with grass-fed cattle from similar herds. The lower emission intensity is associated with faster growth rates and therefore reduced age at slaughter for the grain finished cattle, resulting in lower lifetime enteric methane and manure emissions. Although emissions associated with fossil fuel energy use and crop production are higher for feedlot cattle, the reduced enteric methane emissions more than compensate for this increase. Similar results have been found by Peters et al. (2010a) and Pelletier et al. (2010) for cattle finished in feedlots in Australia and the USA. The impact of including LU and dLUC emissions on this result was quite dependent on the method applied. Using national inventory data where carbon sequestration in existing crop land is reported to be high, there were negligible impacts from LU and dLUC. However, where crop land was assumed to be a net source of emissions, this was a more significant source. Emissions from LU and dLUC sources should be interpreted with caution, because of the high degree of uncertainty in the estimation techniques (Henry et al. 2015). Further research is warranted to understand the magnitude of these emissions in Australian crop lands.

In contrast to the GHG results, we found fresh water consumption to be 49–51% higher and energy demand to be 24–38% higher for grain-finished cattle in the short-fed domestic and mid-fed export market classes. Crop land occupation and stress-weighted water use were also higher than grain-finished cattle from the same region (Wiedemann et al. 2016a). However, total land occupation was 14–24% lower, than grass-finished cattle from the same region. Although the magnitude of impacts per kilogram of LWG in the feedlot were quite different to impacts per kg of LW finished (cradle to gate), the differences between market types were similar for fresh water consumption, energy demand and crop land occupation. However, GHG emissions from the mid-fed export market type were lower than the short-fed domestic market type when the full supply chain impacts were considered, because of the comparatively larger reduction in slaughter age for these cattle and higher finished weights. This suggests that mid-fed cattle with finishing weights of ~650 kg LW are optimal from the perspective of emissions intensity.

Conclusions

Grain finishing was found to have relatively high energy demand, fresh water consumption and crop land occupation, whereas GHG emissions were lower than comparative grass-finished cattle. Emissions intensity was lowest for cattle finished on grain for 125 days to 650 kg LW, suggesting this market type is closest to optimum. The results suggest that increasing turnover weight for the domestic market type would be one approach to improve the environmental efficiency of beef production in Australia. With increasing numbers of cattle finished on grain in Australia, these results suggest lower GHG emissions from cattle produced using these systems, but higher resource use in response to intensification. Further research is required to refine prediction methods for manure emissions, enteric methane in long-fed cattle and emissions from LU and dLUC for different parts of Australia’s crop land. Variation in GHG, energy demand and crop land was largely explained by differences in feed conversion ratio, suggesting that this performance measure is an important indicator of environmental performance. With all other factors remaining equal, continued industry focus on improving feed conversion ratio is expected to yield production benefits and environmental improvements over time in the feedlot industry.

Acknowledgements

This study was funded by Meat and Livestock Australia and completed by FSA Consulting, Toowoomba. The feedlot managers are thanked for supplying data.

References


Grain

Compassion in World Farming
Resource use and impacts from grain-finishing beef


Skerman A (2000) ‘Reference manual for the establishment and operation of beef cattle feedlots in Queensland.’ (Queensland Department of Primary Industries: Toowoomba, Qld)


4.3 ADDENDUM

The results in this paper were modified to enable consistent reporting with other beef and meat production systems by expanding the system boundary to include meat processing, and revising the reference flow/functional unit from ‘kilograms of live weight’ to kilograms of boneless meat (see results in Table 4 and Table 5). Greenhouse gas prediction methods were also standardised to apply the most recent National Inventory Report methods (Commonwealth of Australia, 2016) for enteric methane for the grazing sector of the supply chain. This slightly reduced the predicted enteric methane from 6.5% to the equivalent of 6.3% of gross energy intake (Charmley et al., 2016). Meat processing impacts were modelled using inventory data presented in chapter 6 (Table 3) and meat processing assumptions relating to carcase yield were presented in section 2.2.2 of chapter 6.
5 RESOURCE USE AND ENVIRONMENTAL IMPACTS FROM AUSTRALIAN EXPORT LAMB PRODUCTION: A LIFE CYCLE ASSESSMENT

5.1 ATTRIBUTION STATEMENT

The paper Resource use and environmental impacts from Australian export lamb production: a life cycle assessment was led by S.G Wiedemann and co-authored by M-J Yan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling flocks and impacts from greenhouse gas emissions
- Development and application of novel methods for determining water use from farm storages
- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and flock outputs, and application of novel allocation methods to handle co-production of wool and live weight.
- Primary data acquisition and data analysis
- Preparation of the manuscript and completion of the peer-review process

Yan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with development of figures and tables for the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with analysis and manuscript preparation
5.2 INTRODUCTION

This study was the first multi-impact analysis of lamb production in Australia, and was conducted in parallel with the beef case study research (chapter 3). The study provided the baseline farm analysis for a full supply chain companion study presented in chapter 6.

This research analysed state-wide regional datasets covering the majority of Australia’s prime lamb production systems. The focus on lamb production from meat breed sheep complemented work done by other researchers, and by the author (see chapter 7), in assessing Merino sheep systems. This study applied recently developed biophysical methods to handle allocation (Wiedemann et al., 2015b), providing a preferred approach to handling this issue than the common economic allocation applied in previous studies.

The results provided a robust benchmark and hotspot analysis for the lamb industry, and provided important insights into the effect of high production intensity systems, such as those studied in Victoria, and low production intensity systems, such as those studied in South Australia. Land use and land use change emissions were included, filling an important knowledge gap in this area. The study was also the first to provide detailed, regionalised inventories of crop land and grassland utilised for lamb production.

The water use and water stress assessment study was the first to accurately take into account the effects of farm water supply losses (dams) using validated water balance models, and was the first study to provide an assessment of the impact of irrigation water use in lamb production. In this way, the study expanded and significantly strengthened the knowledge base around water use and water stress for lamb production systems that had been provided by earlier studies (Peters et al., 2010b; Ridoutt et al., 2012b).
Resource use and environmental impacts from Australian export lamb production: a life cycle assessment

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Abstract. This study conducted a life cycle assessment (LCA) investigating energy, land occupation, greenhouse gas (GHG) emissions, fresh water consumption and stress-weighted water use from production of export lamb in the major production regions of New South Wales, Victoria and South Australia. The study used data from regional datasets and case study farms, and applied new methods for assessing water use using detailed farm water balances and water stress weighting. Land occupation was assessed with reference to the proportion of arable and non-arable land and allocation of liveweight (LW) and greasy wool was handled using a protein mass method. Fossil fuel energy demand ranged from 2.5 to 7.0 MJ/kg LW, fresh water consumption from 58.1 to 238.9 L/kg LW, stress-weighted water use from 2.9 to 137.8 L H2O-e/kg LW and crop land occupation from 0.2 to 2.0 m2/kg LW. Fossil fuel energy demand was dominated by on-farm energy demand, and differed between regions and datasets in response to production intensity and the use of purchased inputs such as fertiliser. Regional fresh water consumption was dominated by irrigation water use and losses from farm water supply, with smaller contributions from livestock drinking water. GHG emissions ranged from 6.1 to 7.3 kg CO2-e/kg LW and additional removals or emissions from land use (due to cultivation and fertilisation) and direct land-use change (due to deforestation over previous 20 years) were found to be modest, contributing between −1.6 and 0.3 kg CO2-e/kg LW for different scenarios assessing soil carbon flux. Excluding land use and direct land-use change, enteric CH4 contributed 83–89% of emissions, suggesting that emissions intensity can be reduced by focussing on flock production efficiency. Resource use and emissions were similar for export lamb production in the major production states of Australia, and GHG emissions were similar to other major global lamb producers. The results show impacts from lamb production on competitive resources to be low, as lamb production systems predominantly utilised non-arable land unsuited to alternative food production systems that rely on crop production, and water from regions with low water stress.

Additional keywords: carbon, footprint, GHG, land, water.

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Introduction

Agricultural systems such as lamb production face the challenge of maintaining and increasing production in the future with constrained natural resources and pressure to reduce environmental impacts from production. Globally, meat demand is expected to increase 74% by 2050 because of expanding global population and increased wealth (FAO 2009). However, global targets also exist to reduce greenhouse gas (GHG) emissions (IPCC, Stocker et al. 2013). In Australia, water resources are constrained in major river systems such as the Murray–Darling (ABS 2008), and restricted (capped) supply has led to increased competition between water users (MDBA 2012). Arable land resources are limited by soil type and climate to ~4% of national land mass (Lesslie and Mewett 2013). In this context, resource and environmental efficiency is an increasingly important consideration for agri-food systems such as lamb in order to remain competitive in accessing finite resources. Life cycle assessment (LCA) is an internationally recognised tool for determining whole-supply chain resource use and impacts, accounting for multiple impacts concurrently (ISO 2006). Previous Australian sheep LCA have typically been single impact studies (GHG or water) for a single case study farm (CSF) or a small number of farms, producing predominantly Merino wool-meat sheep (Eady et al. 2012; Brock et al. 2013). Only one study (Ridoutt et al. 2012) investigated prime lamb production and this covered water only, using a case study approach. Peters et al. (2010a, 2010b, 2011) performed the only multi-impact study but this covered only one farm in Western Australia (WA) producing lamb for domestic consumption. In other sectors such as beef, recent LCA studies (Wiedemann et al. 2015a, 2016) have applied a similar multi-impact approach. Our study aimed to address the need for a multi-impact, multi-region analysis of export lamb production to produce a benchmark analysis of resource use and environmental impacts and impact hotspots from cradle to farm gate for Australian export lamb from the three largest production regions. The study used regional datasets to provide a more representative analysis of the study region, and augmented this with detailed case studies to provide specific information regarding on-farm resources and
management. The study applied methods for handling co-production based on protein mass allocation, used farm water supply balances to assess fresh water consumption, expanded the assessment of water use to include the impacts of irrigation and applied a disaggregated land occupation assessment based on land type and suitability for cropping. Additionally, the study conducted original research on land use (LU) and direct land-use change (dLUC) emissions from lamb production.

**Methodology**

**Regions**

The majority of Australian export lambs are drawn from three major production regions: Victoria (Vic.), South Australia (SA) and New South Wales (NSW). Collectively, northern and southern regions in NSW represent 37% of Australia’s sheep flock, Vic. represents 21%, and south-eastern SA represents a further 15% (MLA 2013). In each state, a regional average farm (RAF) was modelled using data from the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES 2013) survey of 203 specialist lamb producers, averaged over 5 years (2006–2010). Additional data were obtained from three to five CSF surveyed in each region in the years 2010–2012. The CSF were located in south-western Vic., the southern and northern tablelands of NSW, and from the Hawker region of SA. Four Vic. CSF farms were selected from the Livestock Farm Monitor Project (DPI 2011) based on lamb sale weight (>45 kg) and breeds to suit the export market. NSW and SA CSF data were obtained from three farms surveyed as part of the study, producing export weight first- or second-cross lambs from meat breed or Merino ewe flocks. A simplified description of the production system, showing the system boundary is provided in Fig. 1.

Impacts were reported on an intensity basis with a functional unit of 1 kg of liveweight (LW) at the farm gate.

**Impacts assessed**

The study investigated GHG emissions using the IPCC AR4 global warming potentials of 25 for CH4 and 298 for N2O (IPCC 2007). GHG emissions associated with LU and dLUC were included and reported separately, following guidance from the Livestock Environmental Assessment and Performance partnership (LEAP 2014). Fossil fuel energy demand was assessed by aggregating all fossil fuel energy inputs throughout the system and reporting these per mega joule (MJ) of energy, using Lower Heating Values. Fresh water consumption was assessed using methods described and the assessment of stress-weighted water use was based on Pfister et al. (2009). Methods for reporting disaggregated land occupation are explained below.

**Production and purchased inputs**

Production data were accessed from detailed farm records, and verified through discussions with farmers at each CSF. Flocks were modelled over the whole production cycle, from an inventory of breeding ewes, replacement ewes, rams and lambs through to the point of sale. Flocks were modelled with static breeder numbers, with replacement breeder numbers being equal to cull breeder numbers and mortalities combined. Replacement ewes were first mated at 18 months of age across all flocks modelled. Critical factors such as weaning and sale weights were cross checked against 2–3 years of data to remove the effect of unusual seasons. The CSF were predominately pasture-based systems. A proportion of lambs are also finished

![Fig. 1. Generalised outline of the lamb production system showing biophysical inputs and co-products. Shaded area indicates the inputs and processes associated with the sheep flock. Cash cropping and beef cattle subsystems were divided from sheep production and co-production of sheep liveweight and wool was handled using biophysical allocation.](image)
with supplementary grain diets in the southern states, and to account for this, two specialist grain-finishing CSF farms were included in Vic. and SA. Supplementary grain feeding was modelled to represent 15% of all lambs finished in these regions. Flock production data are reported in Table 1. Purchased inputs are reported per tonne of dry matter intake (DMI) by the sheep flock (Table 2) as an aggregate indicator of livestock units. These values can be converted to a dry sheep equivalent basis by dividing by 2.5, assuming a standard 400-kg annual DMI per dry sheep equivalent. Transport of livestock and purchased inputs were included, as was staff travel to and from work. Services (i.e. financial, communications, repairs) were modelled based on expenditure using economic input-output data (Rebitzer et al. 2002). Capital infrastructure (i.e. buildings, fences) and machinery were excluded based on their small contribution (<1% of impacts) assessed during the scoping phase. The study required input data at a greater resolution than provided by the ABARES survey for several inputs and productivity factors, requiring additional data and model assumptions. In the absence of ABARES survey data, lamb sale weights were determined from reported lamb sale prices ($/lamb) and market average sale prices ($/kg). Growth rates were determined from lower-bound estimates of lamb age from the corresponding CSF dataset, resulting in growth rates that were intermediate between the CSF and values reported by the DCCEE (2010). Major inputs such as fuel and fertiliser were disaggregated into separate products using the proportions of expenditure from the CSF and market values (available as Supplementary Material for this paper). Impacts generated off-farm via the use of purchased inputs were modelled using background data from the Australian LCI (AustLCI) database where available (Life Cycle Strategies 2007), or the European EcoInvent (2.2) database (Swiss Centre for Life Cycle Inventories 2010). Grains inventory data were modelled using inventory data reported by Wiedemann et al. (2010).

Resource use, emissions and removals

Modelling feed intake

Feed intake was modelled using the AFRC (1990) feed intake model as applied by the Australian National Greenhouse Gas Inventory (DCCEE 2010), which determines intake from LW and feed availability. Feed supplement use was determined from records of purchased inputs, and pasture intake was considered to be the residual of predicted feed intake less supplementary feed inputs.

Land occupation

Land occupation was determined using a disaggregated land inventory that accounted for differences in land type using three categories (measured in m² year): crop land (arable land used to produce crops), arable land used for pasture (land suitable for crop production currently used for pasture, termed arable pasture land) and non-arable land, which is unsuitable for crop production because of limitations to land capability. For the CSF, the proportion of land in each category was determined

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Vic. CSF</th>
<th>Vic. RAF</th>
<th>NSW CSF</th>
<th>NSW RAF</th>
<th>SA CSF</th>
<th>SA RAF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Land occupation</td>
<td>460</td>
<td>453</td>
<td>595</td>
<td>1988</td>
<td>16 061</td>
<td>1096</td>
</tr>
<tr>
<td>Crop land (ha)</td>
<td>23</td>
<td>12</td>
<td>33</td>
<td>9</td>
<td>5</td>
<td>12</td>
</tr>
<tr>
<td>Arable pasture land (ha)</td>
<td>114</td>
<td>22</td>
<td>20</td>
<td>99</td>
<td>0</td>
<td>54</td>
</tr>
<tr>
<td>Non arable land for grazing (ha)</td>
<td>324</td>
<td>419</td>
<td>542</td>
<td>1881</td>
<td>16 056</td>
<td>1030</td>
</tr>
<tr>
<td>Farm dam (%)</td>
<td>73</td>
<td>64</td>
<td>27</td>
<td>73</td>
<td>27</td>
<td>74</td>
</tr>
<tr>
<td>Bore (%)</td>
<td>19</td>
<td>23</td>
<td>23</td>
<td>73</td>
<td>27</td>
<td>74</td>
</tr>
<tr>
<td>River/creek (%)</td>
<td>8</td>
<td>13</td>
<td>0</td>
<td>8</td>
<td>13</td>
<td>0</td>
</tr>
<tr>
<td>Breeding ewes</td>
<td>2364</td>
<td>1309</td>
<td>2220</td>
<td>1255</td>
<td>2733</td>
<td>1171</td>
</tr>
<tr>
<td>Breeding ewe culling rate (%)</td>
<td>21.5</td>
<td>23.0</td>
<td>21.1</td>
<td>23.0</td>
<td>29.0</td>
<td>23.0</td>
</tr>
<tr>
<td>Breeding ewe mortality rate (%)</td>
<td>4.7</td>
<td>3.1</td>
<td>3.2</td>
<td>4.7</td>
<td>4.0</td>
<td>3.2</td>
</tr>
<tr>
<td>Ewe standard reference weight (kg)</td>
<td>67</td>
<td>65</td>
<td>62</td>
<td>65</td>
<td>60</td>
<td>65</td>
</tr>
<tr>
<td>Lambing (% at marking)</td>
<td>111.5</td>
<td>98.6</td>
<td>110.5</td>
<td>91.6</td>
<td>100.8</td>
<td></td>
</tr>
<tr>
<td>Average lamb weight at sale (kg LW)</td>
<td>51</td>
<td>50</td>
<td>50</td>
<td>54</td>
<td>52</td>
<td>49</td>
</tr>
<tr>
<td>Average daily growth rate of lambs (kg/day)</td>
<td>0.18</td>
<td>0.14</td>
<td>0.16</td>
<td>0.13</td>
<td>0.12</td>
<td>0.14</td>
</tr>
<tr>
<td>Annual sheep and lamb sales (kg LW)</td>
<td>134 376</td>
<td>67 312</td>
<td>119 109</td>
<td>66 036</td>
<td>135 598</td>
<td>59 819</td>
</tr>
<tr>
<td>Annual wool sales (kg greasy)</td>
<td>18 263</td>
<td>8185</td>
<td>13 557</td>
<td>8972</td>
<td>29 858</td>
<td>7695</td>
</tr>
<tr>
<td>LW sold per breeding ewe (kg LW/ewe)</td>
<td>56</td>
<td>51</td>
<td>54</td>
<td>53</td>
<td>50</td>
<td>51</td>
</tr>
<tr>
<td>Biophysical allocation for sheep LW (%)</td>
<td>69</td>
<td>72</td>
<td>73</td>
<td>70</td>
<td>59</td>
<td>71</td>
</tr>
<tr>
<td>Economic allocation for sheep LW (%)</td>
<td>81</td>
<td>76</td>
<td>76</td>
<td>69</td>
<td>52</td>
<td>73</td>
</tr>
</tbody>
</table>

A Values reported for the total sheep flock and not allocated between wool and LW.
B Data for the RAF substituted from the CSF dataset.
C Economic allocation was determined from the sale value of sheep and wool, see Supplementary Material.
Table 2. Major purchased inputs of lamb production for the case study farms (CSF) and regional average farms (RAF) from Victoria (Vic.), New South Wales (NSW) and South Australia (SA)

Inputs were expressed per tonne dry matter intake (DMI) for the whole flock.

<table>
<thead>
<tr>
<th></th>
<th>Vic. CSF</th>
<th>Vic. RAF</th>
<th>NSW CSF</th>
<th>NSW RAF</th>
<th>SA CSF</th>
<th>SA RAF</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Feed</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lupins (kg/t DMI)</td>
<td>0.0</td>
<td>5.0</td>
<td>8.0</td>
<td>6.3</td>
<td>0.0</td>
<td>6.5</td>
</tr>
<tr>
<td>Hay (kg/t DMI)</td>
<td>12.7</td>
<td>8.0</td>
<td>0.0</td>
<td>10.0</td>
<td>0.0</td>
<td>10.4</td>
</tr>
<tr>
<td>Cereal grain (kg/t DMI)</td>
<td>6.1</td>
<td>5.0</td>
<td>36.3</td>
<td>6.3</td>
<td>0.0</td>
<td>6.5</td>
</tr>
<tr>
<td>Wheat straw (kg/t DMI)</td>
<td>2.1</td>
<td>3.7</td>
<td>–</td>
<td>–</td>
<td>1.9</td>
<td>4.5</td>
</tr>
<tr>
<td>Feedlot ration (kg/t DMI)</td>
<td>8.2</td>
<td>7.2</td>
<td>–</td>
<td>–</td>
<td>5.0</td>
<td>7.4</td>
</tr>
<tr>
<td><strong>Energy</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Electricity (kWh/t DMI)</td>
<td>2.2</td>
<td>1.7</td>
<td>2.6</td>
<td>2.2</td>
<td>3.3</td>
<td>2.4</td>
</tr>
<tr>
<td>Oil (L/t DMI)</td>
<td>0</td>
<td>0.2</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.2</td>
</tr>
<tr>
<td>Diesel (L/t DMI)</td>
<td>4.5</td>
<td>5.2</td>
<td>1.2</td>
<td>4.8</td>
<td>2.4</td>
<td>7.0</td>
</tr>
<tr>
<td>Petrol/LPG (L/t DMI)</td>
<td>0.3</td>
<td>0.8</td>
<td>0.9</td>
<td>0.8</td>
<td>0.7</td>
<td>1.0</td>
</tr>
<tr>
<td><strong>Fertilisers and soil conditioners</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Superphosphate (kg/t DMI)</td>
<td>19.1</td>
<td>25.0</td>
<td>14.2</td>
<td>18.2</td>
<td>0.0</td>
<td>26.2</td>
</tr>
<tr>
<td>Lime (kg/t DMI)</td>
<td>7.8</td>
<td>6.3</td>
<td>21.4</td>
<td>3.1</td>
<td>0.0</td>
<td>1.1</td>
</tr>
<tr>
<td>Herbicides (g/t DMI)</td>
<td>142.2</td>
<td>303.2</td>
<td>81.6</td>
<td>97.2</td>
<td>0.0</td>
<td>763.4</td>
</tr>
<tr>
<td><strong>Other inputs and services</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Veterinary services ($/t DMI)</td>
<td>9.7</td>
<td>4.3</td>
<td>6.2</td>
<td>5.2</td>
<td>2.8</td>
<td>4.9</td>
</tr>
<tr>
<td>Communication services ($/t DMI)</td>
<td>0.7</td>
<td>0.5</td>
<td>0.5</td>
<td>0.7</td>
<td>0.5</td>
<td>0.7</td>
</tr>
<tr>
<td>Insurance ($/t DMI)</td>
<td>2.1</td>
<td>1.4</td>
<td>1.3</td>
<td>1.9</td>
<td>1.5</td>
<td>2.0</td>
</tr>
<tr>
<td>Accounting ($/t DMI)</td>
<td>1.1</td>
<td>0.8</td>
<td>0.6</td>
<td>1.0</td>
<td>0.7</td>
<td>1.1</td>
</tr>
<tr>
<td>Transport distance (km to service centre)</td>
<td>15</td>
<td>30</td>
<td>20</td>
<td>30</td>
<td>110</td>
<td>50</td>
</tr>
</tbody>
</table>

*AFeedlot ration composition provided in Supplementary Material. Inputs can be converted to whole-flock ‘dry sheep equivalent’ by dividing by 2.5.

from information provided by the farmers during field observations regarding the suitability of paddocks for cultivation, and land areas were verified with an analysis of satellite imagery. For the RAF, crop land was modelled from the inventory of grain and hay inputs. Total pasture land occupation was reported in the ABARES dataset but land capability was not detailed. In lieu of regionally specific data, arable pasture land in the RAF analysis was determined from an analysis of national statistics from the FAO (FAOSTAT 2014) and Lesslie and Mewett (2013), which showed arable land not currently used for cropping to be 4.9% after Wiedemann et al. (2016). No characterisation factors were applied, and land occupation data were reported in square metre years (m² year).

**Fresh water consumption**

The study focussed on fresh water consumption using comprehensive water balance methods recommended by Bayart et al. (2010) to assess uses and losses throughout the foreground and background system. Fresh water consumption refers to evaporative uses or uses that incorporate water into a product that is not subsequently released back into the same river catchment (ISO 2014). Soil stored moisture from rainfall, or so called ‘green water’ was excluded according to ISO (2014). Water consumption associated with historic LUC was conducted in comparison with a reference system of 1990 (Wiedemann et al. 2016) resulting in no change in water yield. This approach provided results that align with the national water accounts (ABS 2012a), which consider water consumption based on current LU rather than in comparison to a historic reference system. Water degradation was not assessed because of a lack of regional characterisation and nutrient transport factors for grazing systems across the regions assessed.

Measured water data on-farm were not available and were modelled from an assessment of water supply and utilisation on the farms using water balances, using methods described in Wiedemann et al. (2016). Sheep drinking water was estimated using the equation determined by Luke (cited in CSIRO 2007).

\[
I_w = 0.1911 \times t - 2.882 \quad (R^2 = 0.84)
\]

Where: \(I_w\) = water intake (L/45 kg LW sheep day); \(t\) = maximum daily air temperature (°C).

Drinking water estimates per sheep were modified to account for liveweight differences using the method outlined by Luke (1987). This clarification was picked up by the authorship since submission. Drinking water consumed by the animal is lost to the atmosphere via respiration and perspiration, integrated into the product and released outside the river catchment or excreted as urine. Each was modelled as a consumptive use. It could be argued that urination results in a flow of water back to the catchment and is therefore not a consumptive use. However, we consider this water to be analogous to irrigation, because it is deposited to pasture in small, dispersed volumes resulting in a high level of evaporation or transpiration. Proportions of drinking water supplied from bores, creeks and rivers or farm dams (Table 1) were determined from the survey and CSF site visits, and were verified by an analysis of water supply points using satellite imagery. Losses associated with water supply were modelled after Wiedemann et al. (2016).
Dam densities were within the range reported by Nathan and Lowe (2012), but extractions for livestock drinking water as a proportion of dam volume were much lower in the present study than the assumptions made by these authors. Farm dam water supply efficiency factors were determined from water balances as an indicator of supply efficiency proportional to total extraction of water from the environment. These factors were 17.5% (Vic. CSF), 18.2% (NSW CSF) and 7.5% (SA CSF). In the absence of farm water data in the ABARES dataset, supply data and dam supply efficiency from the CSF datasets were applied.

Irrigation water use for lamb production was determined from irrigation land reported by the ABARES survey and irrigation rates from the ABS survey of water use on Australian farms (ABS 2012b). None of the CSF used irrigation. Total estimated irrigation water use was verified by comparison with reported irrigation for ‘sheep and other livestock’ for the years 2007/8–2009/10 (ABS 2011). Water use per kg LW from both datasets was in agreement, providing confidence in the estimates based on the ABARES dataset. Irrigation associated with feed grain inputs was modelled using the water inventory of Ridoutt and Poulton (2009) and total irrigation of cereal crops in each state reported by the ABS (2012b). Losses associated with irrigation water supply of 27.1% were applied, based on the ABS national water accounts (ABS 2012b).

Stress-weighted water use

The stress-weighted water use was assessed following Pfister et al. (2009) by accounting for the expected impact of water use in a given catchment using a global stress weighting factor. To calculate the stress-weighted water use, fresh water consumption was multiplied by the regional water stress index (WSI, Pfister et al. 2009) and was then divided by the global average WSI (0.602) and expressed as water equivalents (L H2O-e.) following Ridoutt and Pfister (2010).

Fossil fuel energy demand

Fossil fuel energy demand was associated with both direct energy demand on-farm and energy demand from the manufacture and transport of goods and services used by the farms. Modelling of energy demand was based on the inventory of purchased goods, services (Table 2) and transport distances, using inventory data from AustLCI or EcoInvent (Life Cycle Strategies 2007; Swiss Centre for Life Cycle Inventories 2010).

Greenhouse gas emissions

Emissions from fossil fuel energy were modelled directly from the energy inventory. Livestock GHG emissions (enteric and manure) were modelled using Australian National Greenhouse Gas Inventory methods (DCCEE 2010) (Table 3).

Table 3. Greenhouse gas prediction methods used in the study

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Key parameters/model</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enteric CH4 (kg/sheep)</td>
<td>kg dry matter/head × 0.0188 + 0.00158</td>
<td>DCCEE (2010)</td>
</tr>
<tr>
<td>Manure CH4 (kg CH4/kg dry matter manure)</td>
<td>1.4 × 10⁻³</td>
<td>DCCEE (2010)</td>
</tr>
<tr>
<td>Manure N2O (kg N₂O-N/kg N)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>In urine</td>
<td>0.004</td>
<td>DCCEE (2010)</td>
</tr>
<tr>
<td>In faeces</td>
<td>0.005</td>
<td></td>
</tr>
<tr>
<td>Manure NH3 (kg NH₃-N/kg N excreted in manure)</td>
<td>0.2</td>
<td>DCCEE (2010)</td>
</tr>
<tr>
<td>Indirect N2O from NH3 losses (kg N₂O-N/kg NH₃-N volatilised)</td>
<td>0.004</td>
<td>Derived from DCCEE (2010)</td>
</tr>
<tr>
<td>Indirect N2O from leaching and runoff (kg N₂O-N/kg NH₃-N volatilised)</td>
<td>0.0075</td>
<td>De Klein et al. (2006)</td>
</tr>
</tbody>
</table>

Greenhouse gas emissions from LU and dLUC

Soil carbon (C) changes under crop and pasture soils (LU emissions and removals), were included following guidance from LEAP (2014). Soil C losses from crop land were mainly related to the use of purchased grain sourced regionally. Estimated soil C losses took into account the different rate of loss from conventional and zero-tillage (Dalal and Chan 2001; Chan et al. 2003), assuming multiple cultivations occur on 37% of crop land (NSW, Vic., SA – ABS 2009) with remaining land being zero-tillage. Soil C losses were assumed to be 0.1 t C/ha.year for zero tillage and 0.58 t C/ha.year for land cultivated more than once, with the latter predicted using the equation of Dalal and Chan (2001) for light textured soils (30% clay).

In contrast to cropping soils, soil C sequestration rates of 0.29 ± 0.17 t C/ha.year (15 studies) were reported for Australian pastures, where phosphorus fertiliser and lime have been applied (Sanderman et al. 2010). Chan et al. (2010) reported a C stock change of 9.9 t C/ha over an estimated 25–40 years for fertilised pastures in southern NSW, though such changes have not been reported for all regions (Davy and Koen 2013; Schwenke et al. 2013). Phosphorus fertiliser application was common in the sheep systems studied (Table 2) and C removals were explored using two scenarios: (i) zero change in soil C under fertilised pasture, (ii) or C sequestration of 7.2 t C/ha under fertilised pasture, based on the sequestration rate of 0.29 t C/ha.year in Sanderman et al. (2010) over a 25-year period corresponding to establishment of a new equilibrium soil C level. Carbon removals from sequestration were annualised over a 100-year period to align with the requirements for permanence. The land area fertilised was determined from the total tonnes divided by a standard application rate of 125 kg/ha (NSW and SA) and 150 kg/ha (Vic.) (ABS 2009) with an assumed 3-year fertiliser rotation.

Direct LUC emissions were determined for the previous 20 years (BSI 2011; LEAP 2014) for conversion of forest to grassland or crop land and conversion of grassland to crop land. Although deforestation in southern states of Australia has fallen to very low levels since 1990 (DCCEE 2012), historic emissions must be considered. Conversion of forest to grassland...
attributable to sheep was assumed to be negligible, because of the dramatic decline in Australian sheep numbers from 170 million in 1990 to 68 million in 2010 (ABS 2013), which indicate sheep production is very unlikely to be a driver of expansion of grassland in these regions. In contrast, crop land has expanded by 12%, 21% and 20% for NSW, Vic. and SA, respectively, based on comparison of the maximum reported area of land cultivated for cereal crops in the 5 years before 1990, with the maximum reported area cultivated in the 5 years to 2010 (ABS 2013).

Direct LUC was predominantly from conversion of grassland (DCCEE 2012), assumed to be 80% (NSW) and 95% (Vic., SA) of new crop land, with the remainder arising from conversion of forest land. Total C losses of 84 t C/ha (forest to crop land) and 12.6 t C/ha (grassland to crop land) were assumed using tier II methods (DCCEE 2012), corresponding to annualised emission rates of 15.5 and 2.3 t CO₂-e/ha.year. Divided over the total crop land, dLUC emissions per hectare were 0.61, 0.61 and 0.58 t CO₂-e/ha.year for NSW, Vic. and SA, respectively.

Handling co-production

Following the recommendation of ISO 14044 (ISO 2006), inputs to the farming systems were first divided among farm subsystems (sheep, beef, cropping) and were accounted separately. Inputs that were not specific to a particular subsystem, such as administration overheads or fertiliser inputs to pasture consumed by both sheep and cattle, were divided based on proportion of annual feed intake by each species. Cattle DMI was predicted from herd data in the CSF and ABARES datasets and feed intake prediction methods from the NGGI (DCCEE 2010). System separation methods for the ABARES dataset are described in the Supplementary Material. Within the sheep subproduction system, LW from lambs and culled breeding animals was not differentiated, and impacts associated with the manufacture of an equivalent mass of nylon were subtracted from the impacts from sheep production. Inventory values for nylon were taken from EcoInvent (Swiss Centre for Life Cycle Inventories 2010).

Results

Resource use

Crop land occupation ranged from 0.2 (SA CSF) to 2.0 (NSW CSF) m² year/kg LW (Fig. 2). Crop land was up to 6% of total land occupation with the majority of land being classified as non-arable. Non-arable land occupation varied from 16.7 (Vic. CSF) to 697.4 m²/kg LW (SA CSF) with the intensity of non-arable land occupation corresponding to relative rainfall and pasture production. The semi-arid SA CSF region (697.4 m² year/kg LW) was characterised by large areas of native pasture and very low stocking rates, which is reflected in the unusually high land occupation value.

Fresh water consumption (Table 4) ranged from 58.1 to 238.9 L/kg LW and non-weighted mean values from the CSF across all regions (71.7 L) were one-third of the non-weighted mean RAF results (218.5 L/kg LW). The primary differences between the datasets related to the utilisation of irrigation water and associated supply losses, which did not occur on any CSF but contributed 44%, 72% and 73% for the SA, NSW and Vic. RAF analyses, respectively.

Losses associated with the supply of drinking water ranged from 39.4 to 100.4 L/kg LW. Variability in supply efficiency was influenced by water supply source and net evaporation rates. Higher reliance on dams resulted in higher loss rates compared with bore or creek water supply, and dam supply losses varied with climatic factors such as net evaporation. Livestock drinking water ranged from 12.9 to 29.0 L/kg LW, with differences related primarily to temperature and to a lesser extent productivity.

Total fossil fuel energy demand ranged from 2.5 (SA CSF) to 7.0 (SA RAF) MJ/kg LW (Fig. 2). There was a trend towards higher energy demand for the RAF, which corresponded to higher levels of purchased farm inputs (SA) and on-farm fuel use (NSW) than the corresponding CSF. Farm energy demand from consumption of diesel, petrol and electricity contributed 38–72% of total energy demand (Fig. 2) and was the largest input. When averaged across datasets, the energy demand associated with fertiliser, livestock materials and supplementary feed were very similar, each contributing 10–11% to total energy demand. In contrast, transport was a minor contributor, averaging only 2.3% of total energy demand.

Impacts

Stress-weighted water use differed considerably between regions and datasets, ranging from 2.9 (SA CSF) to 137.8 (NSW RAF) L H₂O-e/kg LW (Table 4). The wide range was mainly the result of large differences in water stress, from very low (parts of Vic., 0.0107) to high (parts of NSW, 0.815). Weighted average values were intermediate between these levels for the RAF. The stress-weighted results suggested lamb produced in NSW had a greater impact on stressed water resources than lamb produced in SA, despite the latter using more water in volumetric terms. However, the WSI is a global index with coarse local resolution, and caution should be applied in drawing conclusions at the state level. GHG emissions (excluding LU and dLUC) ranged from 6.1 (SA CSF) to 7.3 (NSW RAF) kg/kg LW (Fig. 2). The GHG emissions profile was dominated by enteric CH₄ from the sheep flock (83–89%) followed by N₂O and smaller emissions of CO₂ from fossil fuel energy demand and lime application. The two scenarios for LU and dLUC resulted in either a small emission source (ranging from zero to 0.3 kg CO₂ eq/kg LW), where soil C removals were assumed to be zero, or modest removals of −0.9, −1.0 and −1.6 (NSW, Vic. and SA RAF), and to between −0.3 (NSW and
Fig. 2. Fossil fuel energy demand (direct and indirect), greenhouse gas emissions (excluding land use and land-use change) and land occupation per kilogram of liveweight at the farm gate for lamb produced from the case study farms and regional average farms from Victoria, South Australia and New South Wales.
results from the regional analyses (using regional sheep limited and elevated environmental impacts. Weighted mean using an intensity metric presents an effective way to balance the resource use and impacts relative to production.

Investigating resource use and impact intensity

Resource use and impact intensity

Methods applied to handle the co-production of wool and LW can result in large differences in impacts for each product in sheep systems (Wiedemann et al. 2015b). To improve the transparency of the analysis, results are also presented for greasy wool in the Supplementary Material, and sensitivity of allocation method choices were explored using two comparative methods. Economic allocation resulted in 4–14% higher impacts attributed to LW, and associated lower impacts for greasy wool across most cases. In contrast, using system expansion, increased GHG, fresh water consumption, stress-weighted water and land occupation by 37–48%, but reduced fossil fuel energy demand by 254%. This variation between impact categories when applying system expansion relates to the high energy intensity of the avoided nylon product but relatively lower impacts across other categories.

To explore the sensitivity of enteric CH4 assumptions, we remodelled results applying the IPCC 2006 tier II method (Dong et al. 2006), which resulted in a ~4% reduction in GHG in response to the lower emission factor for lambs applied by the IPCC. Emissions or removals from soil C are less well understood than other emission sources for sheep production, with no published sheep LCA addressing these losses to the author’s knowledge. Higher dLUC emissions may exist from crop land because of variation in soil C loss, though this will only have a small impact on lamb because of the small amount of crop land required for lamb production. Alternative assumptions for soil C change under pasture may have a greater effect. If higher levels of soil C sequestration such as the 9.9 t C/ha reported by Chan et al. (2010) could be achieved, removals could be ~35% higher than reported in the ‘high’ sequestration scenario. However, this is unlikely to be observed in lower rainfall and mixed cropping regions (Davy and Koen 2013).

Discussion

Resource use and impact intensity

Investigating resource use and impacts relative to production using an intensity metric presents an effective way to balance the needs for maintained or increased production with the pressure of limited and elevated environmental impacts. Weighted mean results from the regional analyses (using regional sheep flock sizes of Vic., NSW and SA) were 7.1 kg CO2-e, 6.0 MJ, 1.1 m2, 212.2 L and 96.4 L H2O-e/kg LW for GHG (excluding LU and dLUC), fossil fuel energy, crop land, fresh water consumption and stress-weighted water use, respectively. GHG emissions intensity was higher than previous Australian studies that focussed on Merino sheep systems rather than prime lambs, that showed GHG emissions of 5.3 kg CO2-e/kg LW (Eady et al. 2012; Brock et al. 2013). Peters et al. (2010a) reported impacts of 4.4–4.7 kg CO2-e/kg LW for Merino cross-bred lambs in WA. These low values correspond to higher allocation of impacts to greasy wool as each of these studies applied economic allocation to lambs produced from Merino sheep where greasy wool value is high. Although different allocation methods alter results between the two products, this does not alter the overall efficiency of the system; it simply shifts the impacts between the two products. In the studies cited, impacts from wool are higher than reported here indicating the impacts were transferred to the wool product. Improved flock productivity will reduce GHG emissions intensity from animal sources (Alcock and Hegarty 2011), which dominate the emissions profile of lamb. This is achieved by improving flock feed efficiency via the ‘dilution of maintenance’ effect (Johnson et al. 1996). In the present study, emission intensities were lowest from flocks with the highest lamb production per breeding ewe (Vic. CSF) or high lamb and greasy wool production per breeding ewe (SA CSF), suggesting that a focus on either lamb production or dual-purpose lamb-wool can result in low emissions intensity. Inclusion of LU and dLUC emission and removal sources showed these to be a minor source of emissions, adding up to 4% to total GHG in the scenario where no soil C sequestration was assumed to occur under pasture. Where sequestration was assumed to occur, mean emissions in the RAF were 16% lower than when LU and dLUC were excluded. These impacts have not been assessed in previous studies, and the present results suggest that soil C sequestration under fertilised pastures could be a significant removal if achieved, though as noted this is challenging in the Australian context and the results have a high degree of uncertainty.

Fossil fuel energy demand was found to be 37% and 54% lower in the NSW and SA CSF compared with Vic. CSF and differences were observed between farms in the CSF dataset. Energy demand is a function of production intensity and was most strongly related to on-farm fuel used for vehicles and machinery. We found fertiliser manufacture to range from 9% to 15% of energy demand with the exception of the extensive SA CSF where no fertiliser was used. Livestock materials, which were modelled as veterinary inputs using input-output data, were found to be a significant contributor to energy demand, ranging from 7% to 1077

Table 4. Contribution to fresh water consumption and stress-weighted water use per kilogram of liveweight (LW) at the farm gate for lamb produced from the case study farms (CSF) and regional farms (RAF) from Victoria (Vic.), New South Wales (NSW) and South Australia (SA)

<table>
<thead>
<tr>
<th>Water use</th>
<th>Vic. CSF</th>
<th>Vic. RAF</th>
<th>NSW CSF</th>
<th>NSW RAF</th>
<th>SA CSF</th>
<th>SA RAF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Livestock drinking water (L/kg LW)</td>
<td>12.9</td>
<td>13.2</td>
<td>16.3</td>
<td>13.5</td>
<td>22.0</td>
<td>29.0</td>
</tr>
<tr>
<td>Drinking water supply losses (L/kg LW)</td>
<td>42.7</td>
<td>44.3</td>
<td>41.2</td>
<td>39.4</td>
<td>76.2</td>
<td>100.4</td>
</tr>
<tr>
<td>Irrigation (L/kg LW)</td>
<td>0.0</td>
<td>127.0</td>
<td>0.0</td>
<td>113.4</td>
<td>0.0</td>
<td>84.3</td>
</tr>
<tr>
<td>Irrigation supply losses (L/kg LW)</td>
<td>0.0</td>
<td>33.0</td>
<td>0.0</td>
<td>29.5</td>
<td>0.0</td>
<td>21.9</td>
</tr>
<tr>
<td>Other minor uses (L/kg LW)</td>
<td>2.5</td>
<td>2.1</td>
<td>1.0</td>
<td>1.4</td>
<td>0.1</td>
<td>3.2</td>
</tr>
<tr>
<td>Total fresh water consumption (L/kg LW)</td>
<td>58.1</td>
<td>219.6</td>
<td>58.6</td>
<td>197.1</td>
<td>98.3</td>
<td>238.9</td>
</tr>
<tr>
<td>Stress-weighted water (L H2O-e/kg LW)</td>
<td>5.9</td>
<td>86.9</td>
<td>20.7</td>
<td>137.8</td>
<td>2.9</td>
<td>8.8</td>
</tr>
</tbody>
</table>
18%. Production of supplementary feed was also a significant though variable input, contributing from 6% to 23% of on-farm energy demand. Farms that used fodder or grain required higher energy inputs.

The variation in energy demand between individual farms suggested that opportunities may exist to reduce direct energy demand on sheep farms. Interestingly, we found that farms with lower stocking densities corresponded to lower inputs of energy, and did not result in large differences in GHG emissions. This was partly in response to compensatory factors; for example lower energy demand resulted in reduced GHG. However, lower GHG impacts in the SA CSF analysis were also partly related to high wool production and correspondingly high allocation of impacts to wool. Modelled fresh water consumption included assessment of sources not previously considered in Australian studies, including supply losses from farm dams, and irrigation associated with pastures and purchased inputs. Supply losses and irrigation water use dominated fresh water consumption, rather than drinking water. With irrigation removed, water use was similar to the values reported by Peters et al. (2010b) of 64–100 L (converted to LW basis) for lambs produced in WA though this excluded losses associated with water supply and irrigation. Ridoutt et al. (2012) reported water use of 39 L and stress-weighted water use of 14.7 L H2O-e/kg LW for lamb production in Vic. but excluded irrigation, and assumed much lower supply losses from farm dams than our study. The present results highlight the significance of irrigation and supply losses and suggest water use is higher than previously reported. Drinking water is essential for production and cannot be reduced easily. Supply losses from farm dams can be reduced using improved infrastructure (larger and deeper dams) but these strategies are unlikely to be cost effective. Considering the large volume of irrigation water use, this represents a potential target for reducing water use across the industry though reduced feed production from irrigation land must be offset by increases from dryland areas to maintain total production. Trade-offs may also exist between resource use and environmental impacts. Increasing production from dryland areas may require increased inputs from fertiliser to increase pasture production, or utilisation of grain. Productivity improvements can be achieved via genetic improvement and improved animal husbandry, but high quality feed is also a requirement for high reproductive efficiency and growth rates. Methods to produce higher quality feed are often energy intensive (i.e. fertiliser inputs, fuel inputs for cropping) and may utilise different resources such as crop land and irrigation, and hence, comprehensive analyses are required to understand trade-offs. Such trade-offs also exist when considering lamb in the context of other food production systems. For example, some researchers (i.e. Garnett 2011) have promoted replacing red meat with vegetable protein to lower environmental impacts, but the lamb production systems studied here relied predominantly on non-arable land resources, which is unsuitable for producing vegetable protein or feed grain needed as inputs for monogastric meat production systems.

Australian lamb in the global context

The regional analysis showed similar impacts between states for GHG and water, whereas energy and stress-weighted water use varied to a greater extent. Webb et al. (2013) reported emissions of 7.3 kg CO2-e/kg LW (converted from carcass weight) for lamb produced in the UK whereas Ledgard et al. (2011) estimated GHG emissions of 8.6 kg CO2-e/kg LW for NZ lamb. Both results were similar or slightly higher than the average results presented here, and differences in allocation method may explain some variation. Although our study only investigated impacts to the farm gate, Wiedemann et al. (2015c) showed that pre-farm-gate impacts of Australian lamb represent ~89% of impacts through to retail in the USA, which is similar to findings for New Zealand export lamb (Ledgard et al. 2011). Webb et al. (2013) demonstrated that lamb imported from NZ may have lower impacts than lamb produced and sold in the UK market, highlighting that differences at the production level are more significant than export distance. None of the studies reviewed included potential GHG emissions from LU and dLUC. Our analysis suggested that these sources may be a small emission or removal depending on assumptions regarding soil C change under pasture.

Fewer studies have investigated land occupation or fossil energy demand. Total land occupation of non-organic lamb produced in the UK was found to be 6.5 m²/kg LW (converted from carcass weight – Williams et al. 2006). This is significantly lower than the total land occupation of the present study in response to the higher pasture productivity of UK grazing land. Although land occupation was much higher in our study, the majority of land occupation was from non-arable land. In Australia, non-arable land is the largest land resource in the country (Lesslie and Mewett 2013) and from an agricultural perspective is suitable only to ruminant grazing systems.

Reported energy use for lamb production in the UK ranged from 9.7 to 12 MJ/kg LW (converted from carcass weight; Williams et al. 2006; Webb et al. 2013), which tended to be higher than results presented here, particularly in comparison to the CSF dataset. However, these studies included energy demand from renewable sources which were excluded from the present study, which may explain why energy demand was slightly higher in these studies.

Larger differences in water use were observed with studies applying different assessment methods. For example, Mekonnen and Hoekstra (2012) reported virtual water content (VWC) values ranging from 2790 to 5427 L/kg LW (converted to LW basis). Of the total virtual water content, over 90% was green water; soil stored moisture from rainfall in dryland systems. So called ‘blue water’, which is broadly comparable to fresh water consumption in the present study, was 151–279 L/kg LW with a global average of 250 L/kg LW. Supply system losses were not included in these VWC estimates (M. M. Mekonnen, pers. comm.) and these may add additional fresh water consumption to this global analysis depending on the water supply systems used.

Conclusions

This study presents the first multi-criteria resource and environmental impact analysis of farm-gate export lamb production from the three major production regions in eastern Australia. The study applied new methods for handling co-production, estimating water use and GHG from LU and
dLUC, and provided disaggregated land occupation results not previously reported. The analysis of fresh water consumption water use found supply losses and irrigation to be significant losses not previously analysed in detail by lamb LCA. Despite inclusion of additional uses, fresh water consumption was lower than less comprehensive global analyses of blue water use for lamb production. Lamb production used moderate-low amounts of fossil fuel energy and crop land per kilogram of product. Lamb production relied predominantly on non-arable land not suitable for many alternative food production systems that rely directly or indirectly on arable land. GHG emissions were dominated by animal-related emissions, with smaller contributions from CO2 or indirect emission sources. This suggests that improving flock productivity will be most effective in reducing emissions intensity. Soil management was also identified as a potential source of emissions or removals depending on LU, and further research is required to develop a robust understanding of these factors.

Acknowledgements

This study was funded by Meat and Livestock Australia, and could not have been completed without the contribution of data from the many farmers who provided the case studies presented. We thank Jonathan Tocker for his assistance with the Victorian case study farms. Internal reviews by Dr Stewart Ledgard, Dr Beverley Henry and Dr Greg Thoma were appreciated, as was input to project scope from Dr Tom Davison.

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Garnett T (2011) Where are the best opportunities for reducing greenhouse gas emissions in the food system (including the food chain)? *Food Policy* 36, S23–S32. doi:10.1016/j.foodpol.2010.10.010


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5.3 ADDENDUM

The results in this paper were modified to enable consistent reporting with other lamb and meat production systems by expanding the system boundary to include meat processing, and revising the reference flow/functional unit from ‘kilograms of liveweight’ to ‘kilograms of boneless meat’ (see results in Table 4 and Table 5).

Meat processing impacts were modelled using inventory data presented in chapter 6 (Table 3) and meat processing assumptions relating to carcase yield were presented in section 2.2.2 of chapter 6. Retail lamb (bone-in) was converted to boneless by assuming a meat yield of 76% after Wiedemann & Yan (2014).
6 ENVIRONMENTAL IMPACTS AND RESOURCE USE OF AUSTRALIAN BEEF AND LAMB EXPORTED TO THE USA DETERMINED USING LIFE CYCLE ASSESSMENT

6.1 ATTRIBUTION STATEMENT

The paper *Environmental impacts and resource use of Australian beef and lamb exported to the USA determined using life cycle assessment* was led by S.G Wiedemann, and co-authored by E.J. McGahan, C.M. Murphy, M-J. Yan, B.K. Henry, G. Thoma & S. Ledgard. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions, water and land
- Data analysis and modelling of meat processing, including development and application of allocation rules
- Development, analysis and application of the human edible feed conversion ratio (HE-FCR) indicator
- Interpretation and aggregation rules relating to primary data from the papers Wiedemann et al. (2015a) (Beef) and Wiedemann et al. (2015c) (export lamb)
- Preparation of the manuscript and completion of the peer-review process

McGahan contributed to the study and manuscript via:

- Review of the goal and scope phase
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Review of the goal and scope phase
- Assistance with graphs, tables and manuscript preparation

Yan contributed to the study and manuscript via:

- Review of the goal and scope phase
- Assistance with graphs, tables and manuscript preparation

Ledgard contributed to the study and manuscript via:

- Review of the goal and scope phase
• Review of the manuscript

Henry contributed to the study and manuscript via:

• Review of the goal and scope phase
• Interpretation and application of dLUC emission data from the study cited in the paper as Henry et al. (2015c)
• Review of the manuscript

Thoma contributed to the study and manuscript via:

• Review of the goal and scope phase
• Provision of data and unit processes pertaining to shipping and road transport in the USA, and cold storage in the USA
• Review of the manuscript
6.2 INTRODUCTION

This study was the first full supply chain, multi-impact study completed for Australian red meat exports. The study utilised farm-scale data from the studies presented in chapters 3, 4 and 5, together with land use and land use change emission data from the associated study (Henry et al., 2015a). An additional associated study was published outlining methods and inventory data for meat processing (Wiedemann and Yan, 2014), see supplementary materials.

The study aimed to fill a knowledge gap around the important USA export market, and particularly to address misinformation regarding the environmental impact of food miles and promotion of ‘local food’ for environmental sustainability reasons. The study showed that transport is only a minor contributor to GHG emissions for exported red meat. While the study did not include a comparison with beef or lamb produced in the USA within the scope of the study, it was able to provide a transparent report about Australian production for interested stakeholders.

The study extended the supply chain to include all aspects of meat processing and reported the results using a ‘retail ready’ reference flow (functional) unit. This improved on the previous study of Peters et al. (2010a) which reported results relative to hot standard carcase weight (HSCW) and other Australian studies which had focused on the farm gate segment of the supply chain only.

The study also included an analysis of human edible protein conversion efficiency (HEP-CE) which was an important new indicator incorporated to highlight the role of ruminant systems in producing protein for human consumption from non-edible sources (grass).

This study applied a novel approach where allocation of meat and human edible offal was avoided by considering these outputs as functionally equivalent from a human nutrition perspective. This novel approach was adopted by the international livestock LCA guidance (see LEAP, 2014; LEAP, 2015a; LEAP, 2016).
Environmental impacts and resource use of Australian beef and lamb exported to the USA determined using life cycle assessment

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Abstract

Australia is one of the two largest exporting nations for beef and lamb in the world and the USA is a major export market for both products. To inform the Australian red meat industry regarding the environmental performance of exported food products, this study conducted the first multi-impact analysis of Australian red meat export supply chains including all stages through to warehousing in the USA. A large, integrated dataset based on case study farms and regional survey was used to model beef and lamb from major representative production regions in eastern Australia. Per kilogram of retail-ready red meat, fresh water consumption ranged from 441.7 to 597.6 L across the production systems, stress-weighted water use from 108.5 to 169.4 L H2O-e, fossil energy from 28.1 to 46.6 MJ, crop land occupation from 2.5 to 29.9 m2 and human edible protein conversion efficiency ranged from 7.9 to 0.3, with major differences observed between grass-finished and grain-finished production. GHG emissions excluding land use and direct land use change ranged from 16.1 to 27.2 kg CO2-e per kilogram, and removals and emissions from land use and direct land use change ranged from –2.4 to 8.7 kg CO2-e per kilogram of retail retail ready meat.

Process based life cycle assessment shows that environmental impacts and resource use were highest in the farm and feedlot phase. Transportation contributed ≤5% of greenhouse gas emissions, water and land, confirming that food miles is not a suitable indicator of environmental impacts for red meat transported by ocean shipping. The contribution of international transportation to total energy demand was higher, ranging from 14 to 23%. These beef and lamb supply chains were found to rely on small volumes of water from stressed water catchments, and occupied only small amounts of crop land suited to other food production systems. Production of high quality protein foods for human consumption used only small amounts of protein from human edible grain.

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1. Introduction

Agricultural systems such as livestock production face the challenge of maintaining and increasing production in the future with constrained natural resources and pressure to reduce environmental impacts. Globally, meat demand is expected to increase 74% by 2050 because of expanding global population and increased wealth (FAO, 2009). However, global targets also exist to reduce greenhouse gas (GHG) emissions (Stocker et al., 2013) and concerns exist regarding the use of scarce water resources (Rockström et al., 2007; WHO, 2009) and arable land (UNEP, 2014). Australia is one of the two largest exporting nations for beef and lamb in the world, closely following Brazil in total volume of beef and New Zealand in lamb (FAO, 2011). The United States of America (USA) is a major export market for both products (DFAT, 2012). Product life cycle assessment (LCA) is an important method for understanding the impacts associated with food products and particularly for determining what stages in the supply chain contribute to impacts. Despite long transport distances, LCA studies of red meat have shown that transportation distance, or ‘food miles’ (Paxton, 2011) is not a good indicator of environmental impacts in several instances (Webb et al., 2013; Weber and Matthews, 2008). However, ‘food
miles' is still taken as a proxy indicator for environmental impact in popular media communications and a greater understanding of the relationship between impacts and transport distance is sought by the users of Australian beef and lamb in the USA.

To date, there has been no holistic environmental analysis of Australian beef and lamb supply chains to the USA. Life cycle assessment studies of Australian production have focussed on case study farms (Eady et al., 2011; Peters et al., 2010a, 2010b), theoretical production systems (Ridoutt et al., 2012a) or controlled production systems found on research farms (Brock et al., 2013) that could not be considered representative of markets that draw from large production regions. These studies predominantly focussed on one or two impacts only. Recent farm gate studies of beef (Wiedemann et al., 2015c) and lamb (Wiedemann et al., 2015d) cover larger regions representative of Australia's export markets through to the farm gate, and were the basis for this expanded supply chain analysis. The present study aimed to determine major environmental impacts and resource use from the production, processing and transport of Australian beef and lamb to the USA by extending two existing farm-gate LCA studies by the same authors, which used large, integrated datasets based on case study farms and regional survey datasets. The study aimed to report on major environmental impacts and resource use indicators with new methods and to provide a robust assessment of impacts and hotspots in the supply chain, with particular attention to the role of transportation.

2. Materials and methods

The study included beef and lamb production from major representative production regions in eastern Australia, through the whole supply chain to the point of distribution to retail in the USA. The functional unit was chosen as one kilogram of retail ready cuts of Australian beef and lamb, at the regional storage centre in the USA. The system boundary included all stages of production, processing, transport and cold storage on the east coast of the USA, as well as distribution to the point of retail (Fig 1).

2.1. Production system characteristics

Australia’s sheep and cattle industries have been developed to utilise some 3.5 M km² of native vegetation grazing land (Lesslie and Mewett, 2013), or 46% of continental land area. The majority of sheep are produced in the south-east states of South Australia (SA), Victoria (VIC) and New South Wales (NSW), representing 73% of Australia’s sheep flock (MLA, 2013a) and the vast majority of export lamb production. The majority of beef cattle are produced in the states of Queensland (QLD) and NSW. Central and southern QLD, and northern and central NSW represent 35% of Australia’s beef herd and the major regions exporting premium beef to the USA market and were the focus of the study. Premium beef and lamb exports to the USA must meet specific market requirements. Export lambs are >22 kg carcase weight (CW) and beef cattle destined for premium markets may be grass-fed or grain-finished. The study investigated beef bred in rangeland areas and finished on pastures (grass-fed), and steers finished on grain for either 115 days (Mid-fed — MF) or 330 days (Long-fed — LF). The LF category is tailored to the production of a high quality, niche beef product, predominantly from Angus or Wagyu breeds, for the USA restaurant trade.

2.1.1. Indicators

The study investigated GHG emissions using the IPCC AR4 global warming potentials (GWP, 100 years) of 25 for methane and 298 for nitrous oxide (IPCC, 2007). Greenhouse gas emissions associated with land use (LU) and direct land use change (dLUC) were included and reported separately. Fossil fuel energy demand was assessed by aggregating all fossil fuel energy inputs throughout the system and reporting these per mega joule (MJ) of energy, using Lower Heating Values (LHV). Modelling methods and processes used are described below. Fresh water consumption (an inventory method — Bayart et al., 2010) was assessed, covering all sources and losses associated with livestock production in both foreground and background systems. Fresh water consumption refers to evaporative uses or uses that incorporate water into a product that is subsequently released back into the same river catchment (ISO, 2014). Stress-weighted water use was assessed using the water stress index (WSI) of Pfister et al. (2009) reported in water equivalents (L H2O-e) after Ridoutt and Pfister (2010). Land occupation was reported using a disaggregated inventory based on land type and suitability. Four land

List of acronyms:

<table>
<thead>
<tr>
<th>Acronym</th>
<th>Description</th>
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<tbody>
<tr>
<td>ABARES</td>
<td>Australian Bureau of Agricultural and Resource Economics and Sciences</td>
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<td>CSF</td>
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<td>greenhouse gas</td>
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![Fig 1. Illustration of beef and lamb supply chains in the study. Shaded boxes indicate co-products.](image-url)
types are specified, crop land, arable pasture land, non-arable pasture land and industrial land occupation. No characterisation factors were applied, and land occupation data were reported in square metre years (m² yr). Human edible protein conversion efficiency was determined by dividing the protein content of the retail ready product by human edible protein in the feed consumed and methods are explained in more detail in the following sections.

2.2. Life cycle inventory and modelling
The inventory stage was divided into segments covering the production phase in Australia, the meat processing stage in Australia, and transport and warehousing through the supply chain to the USA. Data collection and modelling methods are outlined for each stage.

2.2.1. Farm and feedlot
Inventory data covering livestock production, inputs of purchased feed, fertiliser, fuel and services, and land occupation for the farm stage of the supply chain were from regional survey data collected by the Australian Bureau of Agricultural and Resource Economics (ABARES) and a survey of case study farms (CSI). The ABARES dataset covered 345 beef producers and 203 specialist lamb producers annually for the five year period of 2006–2010 (ABARES, 2013a, b). Case study farm data were sourced from other publications (Wiedemann et al., 2015c, 2015d). Feedlot inventory data were from a survey of the Australian feedlot industry (Davis et al., 2010a, b). In addition to purchased inputs, Davis et al. (2010a) provided an inventory of measured water use and feed requirements for feedlot finishing. Material inputs for the farm and feedlot are provided in the supplementary material. For each region, human edible protein consumed by livestock was determined from the inventory of grain use throughout the supply chain based on the mass of human edible grain for each grain type, and the average protein content of the grain. Impacts associated with farm infrastructure were excluded based on the findings of a scoping study showing the contribution from these sources was <1% (Wiedemann, unpublished data). Inventory data were aggregated for reporting relative to livestock feed intake.

For each major region, a regional average farm model was developed based on the herd and flock inventory data, which determined total production outputs from the farm stage. This model was also used to determine total feed requirements (dry matter intake – DMI) for the grazing herds and flocks in each region, using the prediction equation of Minson and McDonald (1987) for cattle, and the AFRC (1990) feed intake model for sheep as they are applied in the Australian NGGI (DCCEE, 2012a). Results from the regional herds and flocks were aggregated to provide a market average for the USA export market with ratios for beef of 65% from NSW and 35% from QLD for the grass-fed and mid-fed grain finished supply chains. The long-fed supply chain was only supplied by NSW where a greater number of suitable cattle are bred. Lamb exports were modelled with equal proportions of lamb fed grain and finishing. Material inputs for the farm and feedlot are provided in the supplementary material. For each region, human edible protein consumed by livestock was determined from the inventory of grain use throughout the supply chain based on the mass of human edible grain for each grain type, and the average protein content of the grain. Impacts associated with farm infrastructure were excluded based on the findings of a scoping study showing the contribution from these sources was <1% (Wiedemann, unpublished data). Inventory data were aggregated for reporting relative to livestock feed intake.

Each regional farm model estimated livestock drinking water based on livestock numbers, feed intake and climate. Farm water supply losses and irrigation water use were determined for each region using water balance methods and datasets described in detail by Wiedemann et al. (2015a, 2015b). For each regional farm, a disaggregated model of land occupation was developed to differentiate between crop land, arable pasture land and non-arable pasture land, using methods described in other publications Wiedemann et al. (2015a, 2015b).

Impacts associated with purchased inputs such as fertiliser and fuel were modelled using processes from the Australian LCI database (Life Cycle Strategies, 2007) where available, or the European Ecoinvent (2.0) database (Frischknecht et al., 2005). Feed grain inventory data were obtained from Wiedemann et al. (2010a) and Wiedemann and McGahan (2011). Impacts associated with services such as communications, insurance and accounting were modelled based on expenditure using economic input–output data (Rebitzer et al., 2002).

2.2.2. Meat processing
Inputs and impacts associated with beef and lamb processing (Table 3) were collected from an industry survey of meat processing plants in Australia (GHD, 2011) which included recorded energy inputs, cleaning inputs, water use and waste water production per unit of output.

Two meat processing plant models were developed to determine flows of co-products and waste from processing of beef and lamb, and to determine emissions from waste treatment. Factors used to determine product mass, co-product mass and waste production are described in Wiedemann and Yan (2014). Key model parameters included dressing percentages (i.e. from live animal to hot carcass) of 55% and 45%, cutting and chilling losses of 3% and 4%, and retail yield (from cold carcass to retail meat) 74% and 88% for boneless beef and bone-in lamb respectively. Human edible protein conversion efficiency was determined by dividing the mass of protein in boneless beef and bone-in lamb, determined from the edible yield of each product (Wiedemann and Yan, 2014) and a dry mass protein content of 0.2 kg/kg red meat. Protein mass was therefore 0.19 and 0.15 kg/kg retail product for beef and lamb respectively, and these values were used to determine by the total human edible protein yield. Emissions from anaerobic treatment of effluent were included using effluent production rates and characteristics reported by GHD (2011) and from data collected at three meat processing plants by the authors, and Australian inventory factors (DCCEE, 2012a).

<table>
<thead>
<tr>
<th>Table 1</th>
</tr>
</thead>
<tbody>
<tr>
<td>Grass-fed beef</td>
</tr>
<tr>
<td>Breeding animals (no. cows/ewes)</td>
</tr>
<tr>
<td>Breeding animal culling rate (%)</td>
</tr>
<tr>
<td>Breeding animal mortality rate (%)</td>
</tr>
<tr>
<td>Breeding animal average weight* (kg LW/head)</td>
</tr>
<tr>
<td>Weaning percent (%)</td>
</tr>
<tr>
<td>Weaning weight (kg LW)</td>
</tr>
<tr>
<td>Average steer weight at sale (kg LW/head)</td>
</tr>
<tr>
<td>Average lamb weight at sale (kg LW/head)</td>
</tr>
<tr>
<td>Average daily growth rate of steers/lambs (kg LW/head/day)</td>
</tr>
<tr>
<td>Annual cattle sales (kg LW)</td>
</tr>
<tr>
<td>Annual sheep and lamb sales (kg LW)</td>
</tr>
<tr>
<td>Annual wool sales (kg greasy)</td>
</tr>
<tr>
<td>Biophysical allocation for sheep LW (%)</td>
</tr>
</tbody>
</table>

* LW – live weight, n.a. – not available, n.r. – not reported.
Table 2: Greenhouse gas (GHG) methods and uncertainty estimates used in the study.

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Key parameters/model</th>
<th>Uncertainty (%)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Enteric methane (kg/sheep)</td>
<td>kg DMI/head × 0.0118 + 0.00158</td>
<td>±20</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Enteric methane (CH4 yield – % of GEI grazing cattle)</td>
<td>6.5%</td>
<td>±20</td>
<td>Dong et al. (2006).</td>
</tr>
<tr>
<td>Manure methane (grazing) (kg)</td>
<td>kg DMI* (1 – MEF)</td>
<td>±20</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Manure nitrous oxide (grazing)</td>
<td>Faecal N – 0.005 kg N2O–N/kg N in faeces.</td>
<td>±50</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Manure ammonia (grazing)</td>
<td>0.2 kg NH3–N/kg N of excreted in manure</td>
<td>±20</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Manure methane (kg/bd feedlot beef)*</td>
<td>VS (kg/head)* Bo* MCP p</td>
<td>±20%</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Manure nitrous oxide (feedlot cattle)</td>
<td>Faecal and urinary N – 0.01 kg N2ON/kg N excreted</td>
<td>±50%</td>
<td>Muir (2011).</td>
</tr>
<tr>
<td>Manure ammonia (feedlot cattle)</td>
<td>0.75 kg NH3–N/kg N excreted</td>
<td>±20%</td>
<td>Watts et al. (2012).</td>
</tr>
<tr>
<td>Indirect nitrous oxide – ammonia</td>
<td>0.01 kg N2O–N/kg N lost as ammonia-N</td>
<td>±50</td>
<td>DCCEE (2012a).</td>
</tr>
<tr>
<td>Indirect nitrous oxide – Leaching and runoff</td>
<td>0.0125 kg N2O–N/kg N lost in leaching and runoff</td>
<td>±50</td>
<td>DCCEE (2012a).</td>
</tr>
</tbody>
</table>

a DMI – Dry Matter intake.
b GEI – Gross energy intake assumed to be 18.6 MJ/kg dry matter intake (DMI).
d MEF – Dry matter digestibility; MEF = Methane emission factors (temperate = 1.4 × 10−5 kg CH4/kg DM manure, warm = 5.4 × 10−5 kg CH4/kg DM manure).
e V/S – volatile solids; Bo = 0.17 m3 CH4/kg VS; MCF = 0.510*SR; MCF = 1.736*C/C2; MCF = 2.648*C/C19.

2.2.3. Transport and warehousing

Transport stages were included throughout the supply chain based on an inventory of transport distances and representative truck types and load specifications (Table 4). International transport of chilled beef and lamb was via ship to the USA. Transport distances represented export from the port of Brisbane and import to the port of Philadelphia (PH) based on trade data (https://usatrade.census.gov/). Alternative modelling to the port of Los Angeles was completed for comparison. An inventory of inputs associated with storage in refrigerated warehouse was based on micro data from the Energy Information Agency Commercial Buildings Energy Consumption Survey, ASHRAE design guidelines (ASHRAE, 2004; EIA, 2003), and an interview of plant managers to determine electricity consumption, conducted as part of the project (Table 4).

Impacts associated with transport were modelled for each different transportation type using database processes corresponding to the specific inventory stage. Australian truck processes were modelled using the AustLCI database (Life Cycle Strategies, 2007). International ocean liner transport was modelled using data from Webb et al. (2013). GHG emissions from refrigerant losses were not included because of a lack of data and their expected minor contribution (Webb et al., 2013). Ships were assumed to carry goods both ways because Australia operates a large trade deficit with the USA (DFAT, 2012). For transport within the USA, US lifecycle inventory (National Renewable Energy Laboratory, 2012) long-haul truck processes were applied. To understand the sensitivity of modelling assumptions in transportation, baseline parameters (Table 4) used to model the transportation from Australia to the USA and warehousing in USA were varied as part of the sensitivity analysis.

2.2.4. Methods for handling co-production

The study applied allocation methods following the (ISO, 2006) hierarchy and guidance from (LEAP, 2014) where clear and suitable methods were available. System subdivision and allocation prior to the farm-gate was explained in Wiedemann et al. (2015c, 2015d), using a system separation where possible, and biophysical allocation between wool and live weight in the sheep system after Wiedemann et al. (2015b).

At the meat processing plant, raw hides, tallow, meat and blood meal are co-generated. Allocation between meat products and co-products at the point of meat processing was handled using economic allocation (LEAP, 2014) based on Australian market data (MLA, 2013b, c) and allocation fractions are provided in Table 5.

To understand the sensitivity of the co-product handling at meat processing, an alternative, hybrid approach was analysed. In this approach, we divided the rendering process from the rest of meat processing system. Biophysical allocation was used between retail meat, edible offal and raw hides, and system expansion was applied to determine allocation factors and system expansion factors and methods is described in Wiedemann and Yan (2014).

2.4. Analysis

Modelling and the uncertainty analysis was conducted in Simapro 7.3 (Pré-Consultants, 2012). Uncertainty associated with purchased inputs was determined using a pedigree matrix (Frischknecht et al., 2005), and was assessed using Monte Carlo analysis in SimaPro 7.3. One thousand iterations provided a 95% confidence interval for the results. Differences between datasets were assessed using comparative Monte Carlo analysis in SimaPro 7.3.

3. Results

Results are presented for resource use and impact categories of retail ready beef and lamb in the following sections.
3.1. Fresh water consumption and stress-weighted water use

Fresh water consumption ranged from 441.7 to 597.6 L/kg boneless beef depending on the production system and averaged 463.8 L/kg bone-in lamb (Fig 2). Drinking water and irrigation were the largest sources of fresh water consumption. With supply losses included, drinking water supply contributed 63 and 56% of fresh water consumption, and irrigation contributed 31 and 61% for the average of the beef and lamb supply chains respectively. Water supply losses were much higher than livestock drinking water due to the large evaporative loss from farm dams, which were the major drinking water supply. Irrigation losses were lower than irrigation water use due to the major supply source being larger, more efficient supply dams or bores. Stress-weighted water use ranged from 108.5 to 124.9 L-e/kg bone-in lamb. Stress-weighted water use was strongly influenced by regional water stress indexes, which averaged 0.22 (0.02–0.85) and 0.37 (0.01–0.82) in the beef and lamb production regions. This resulted in higher stress-weighted water use for lamb than beef, despite the higher average fresh water consumption for the latter.

3.2. Land occupation

Total land occupation varied from 199.4 to 432.5 m²/kg boneless beef, 88% of which was from non-arable land (Fig 2). Crop land occupation from the grass and grain finished beef supply chains varied from 3.2 to 29.9 m²/kg boneless beef, with the small amount of crop land used for grass-fed beef contributed by forage or supplement production. Grain production was the predominant contributor to crop land occupation. Consequently, Beef MF and LF occupied 5.1 to 9.5 times more crop land than grass finished beef. Crop land occupation from the lamb supply chain averaged 2.5 m²/kg bone-in lamb.

3.3. Fossil fuel energy demand

Total fossil fuel demand from the grass and grain finished beef supply chains ranged from 32.3 to 46.6 MJ/kg boneless beef (Fig 2). The largest contribution was from the farm and feedlot stage (averaging 63%), followed by meat processing (averaging 20%), and international transportation (predominantly shipping) of meat from Australia to the warehouse in the USA (averaging 17%). Fossil fuel demand associated with transportation in the production system contributed a small amount (1–2%) of total energy demand. Grain-finished supply chains (Beef MF and LF) were found to have higher energy intensity than the grassland-finished supply chain. Total fossil fuel energy demand for the lamb supply chain averaged 28.1 MJ/kg bone-in lamb (Fig 2). The largest contribution was from the farm production stage (46%), followed by meat processing (31%) and international transportation (23%).

3.4. Greenhouse gas emissions

Emissions (excl. LU, dLUC) from the grass and grain finished beef supply chains ranged from 23.4 to 27.2 kg CO₂-e/kg beef. The predominant contribution was from primary production (averaging 93%), followed by meat processing (4%) and transportation of meat from Australia to the warehouse in the USA (3%). By source, enteric methane was the single largest emission (averaging 70%), followed by carbon dioxide from the burning of fossil fuels (averaging 11%), and manure nitrous oxide (averaging 10%). Total GHG emissions from lamb supply chain averaged 16.1 kg CO₂-e/kg bone-in lamb. Relative contributions of components were similar between the beef and lamb supply chains. Lamb primary production was also the predominant contributor (90%) and 75% of emissions were from enteric methane. Contributions from carbon dioxide were slightly higher (14%) than manure nitrous oxide (9%) was slightly lower for lamb GHG emissions relative to beef emissions.

Mean LU and dLUC emissions from the low and high emission scenarios were 8.3, 8.7 and 4.1 kg CO₂-e/kg boneless beef for grass-
fed, mid-fed and long-fed grain finished beef respectively when soil organic carbon stock change was assumed to be zero under pasture. When soil carbon sequestration was included for pastures for beef produced in NSW, values declined slightly to 7.3, 8.1 and 3.5 kg CO₂-e/kg boneless beef for grass-fed, mid-fed and long-fed beef respectively. Results from the two scenarios for lamb showed average emissions of 0.4 kg CO₂-e/kg bone-in lamb, or removals of ~2.4 kg CO₂-e/kg lamb where soil carbon sequestration under improved pastures was included.

3.5. Human edible protein conversion efficiency

Total HEP-CE was 7.9 for grass finished boneless beef and 2.9 for bone-in lamb, indicating that these products yield more human edible protein than they utilise throughout the system via consumption of animal feeds that could potentially be fed to humans. Grain finished boneless beef utilised more human edible protein inputs, resulting in lower efficiencies of 0.3 (LF) and 0.5 (MF).

4. Discussion

Livestock production, processing and transport requires water and energy inputs, and utilises land resources. While a number of studies have investigated resource use at the farm-gate level (Ridoutt et al., 2012a; Wiedemann et al., 2015c, 2015d; Williams et al., 2006), fewer have included a detailed analysis of meat processing and the associated allocation processes required. We found the primary production phase of the supply chain to dominate GHG emissions, fresh water consumption and land occupation, while fossil fuel energy demand was more evenly distributed across the supply chain.

The farm-gate component of this study extended the assessment of fresh water consumption by taking into account consumptive losses associated with water supply throughout the supply chain, accounting for more loss pathways than previously included by some authors (e.g. Peters et al., 2010b). Farm-gate water use was slightly lower than reported for the national average for Australian beef (Wiedemann et al., 2015a), reflecting differences between the specific regions supplying the USA prime beef market and a broader assessment of Australian beef production. Several published studies from the United States do not report their coverage of the supply chain, but report higher water use of ~2300 L/kg boneless beef (converted from carcase weight — Capper, 2011) to 3682 L/kg boneless beef (Beckett and Oltjen, 1993). Comparative stress weighed water use was found to be much lower than fresh water consumption, reflecting the relatively low water stress conditions existing in most Australian livestock producing regions. Similar findings were made for Australian lamb by Ridoutt et al. (2012b).

We found Australian livestock production to rely predominantly on non-arable rangelands, reflecting the land capability in...
Australia, where less than 6.5% of land mass is considered arable (FAOSTAT, 2014). While total land occupation in the study was higher than found by others (e.g. Nguyen et al., 2010; Williams et al., 2006), comparisons are not meaningful without understanding land capability or disturbance. Further research investigating biodiversity impacts is required to determine impacts from land occupation more accurately, but at a minimum differentiating the use of crop land (high disturbance) from pasture land (low disturbance) is informative. Production of the major grass-finished beef and lamb exports was found to utilise very little grain, yielding more HEP than consumed by the animals. This study found energy intensity per kilogram of retail ready meat to be lower from grass finishing than grain finishing. The higher energy intensity from grain finishing primarily relates to the additional inputs required to produce, transport and mill the feed inputs. In comparison, the grass finishing systems used few inputs for pasture production. Together with the land occupation results, this finding shows the importance of ruminant livestock in producing food products from low value land and grasses unsuitable for alternative food production. This is particularly relevant for Australia, where the area of arable land is small comparative to the area of natural rangelands.

Greenhouse gas emissions are known to be dominated by the production phase of the supply chains for red meat (Ledgard et al., 2011). Results from the production phase were similar to previous Australian studies of beef (Eady et al., 2011; Peters et al., 2010a) and lamb (Brock et al., 2013; Eady et al., 2012) when GWP values and enteric methane prediction models were standardised. Impacts from lamb were lower than beef at the farm gate predominantly because a proportion of impacts are allocated to wool (see supplementary material) and because the sheep systems had higher productivity, resulting in lower livestock emissions. Emissions from Australian beef excluding LU and dLUC were similar to slightly lower than results from suckler beef production in Europe (e.g. Casey and Holden, 2006; Nguyen et al., 2012; Williams et al., 2006) and North America (e.g. Beauchemin et al., 2010; Lupo et al., 2013; Pelletier et al., 2010). This is mainly due to lower energy inputs on Australian cattle farms, and lower nitrous oxide emissions from cropping and manure in Australia than international defaults (DCCEE, 2012a). Differences in the impacts from beef and lamb diminished when the different yield of edible product (Table 5) was taken into account; per kg of edible product, GHG emission intensities ranged from 24.5 to 26.8 per kg beef and 21.1 per kg lamb.

Fewer studies have assessed impacts from LU and dLUC sources though these may be significant. Estimation and attribution of emissions and removals is complex, and results were regionally variable, contributing 4.1–8.7 kg CO₂–e/kg beef. Removals or modest emission levels from the sheep system ranged from −2.4 to 0.4 kg CO₂–e/kg lamb, in response to modest rates of soil carbon sequestration under pastures in southern Australia. Emissions from dLUC have declined rapidly in Australia (DCCEE, 2012b) in response to changed management practices and legislation introduced over the last decade. These changes are only realised slowly when applying a retrospective analysis and 20-year amortization. We explored the expected future emissions from Australian beef by projecting the clearing rate data forward over the period from 2006 to 2026, when the full impact of reduced deforestation will take effect. This showed that dLUC emissions for Australian beef will decline to between 0.4 and 0.7 kg CO₂–e/kg beef, with potential for reforestation on previously cleared land to result in net removal of carbon dioxide in this time period (Henry et al. submitted). Rates of GHG removals associated with increased soil carbon levels in improved pasture for the lamb production system are expected to decline as a new equilibrium is reached. Reported dLUC in the literature range from 5 to 8 kg CO₂–e/kg for dairy calves produced in the EU (Nguyen et al., 2010) to >700 kg CO₂–e/kg beef in Brazil (Cederberg et al., 2011). In contrast to these results, Pelletier et al. (2010) investigated soil carbon sequestration in grazing systems and suggested that removals may be up to 8 kg CO₂–e/kg beef for cattle raised on improved pastures in the USA, though a complete investigation of LU and dLUC emissions and removals was not included. Considering the limited data within the LCA literature, further research is required to understand the role of LU and dLUC on beef production.

4.1. Handling co-production

Processing and transport are minor stages and are occasionally overlooked in studies of beef and lamb. Results from a number of recent and older studies (e.g. Lupo et al., 2013; Williams et al., 2006) report results using a carcass weight functional unit without including impacts or allocation processes involved in meat processing, showing a mismatch between the system boundary and reference flow or functional unit of the study (Wiedemann and Yan, 2014). By accounting for the value of co-products and impacts from the meat processing stage, impacts to grass-fed beef, for example, were in the order of 10% lower than if co-products were ignored in the present study. To understand the sensitivity of economic allocation at the meat processing, a hybrid approach was also performed to divide impacts between the primary meat products while using system expansion to account for the minor rendering products. While it has been accepted for some time in LCA practice that multiple methods of allocation may be used at different points in the same study, here this was applied for a closely related co-product system coming from the same process. The hybrid approach resulted in an average of 8% lower GHG emissions, fossil fuel use, fresh water consumption, and stress-weighted water use, and 21% lower crop land occupation for the primary products from beef and lamb supply chains. This highlighted the sensitivity of allocation at meat processing and the contribution of valuable co-products from meat processing.

4.2. Transportation

When averaged across the beef and lamb supply chains, international transportation contributed contributed ≤5% to GHG emissions, land occupation and fresh water consumption, while the contribution to energy demand was higher, ranging from 14% to 20% for beef and 23% for lamb. The higher relative contribution for lamb was related to the lower energy intensity at farm gate (Fig 2). To test assumptions, we investigated alternative transport distances to port and within the USA. Importing beef and lamb into the closer port (Los Angeles) reduced GHG by 0.3% and a modest impact on energy use (−4%). Increasing the distance of post-warehouse transport from 600 km to 2000 km increased the overall energy use by 6–7%, while changes to GHG were −1%. The minor role of transport in assessing impacts from globally traded red meat has been shown previously (Ledgard et al., 2011; Webb et al., 2013) where transport contributed similar levels to GHG and energy as found here. Thus, transport distance or ‘food miles’ was found to be a poor predictor of the environmental impacts and resources of red meat transported by ship.

5. Conclusions

This study is the first multi-impact analysis of Australian red meat supply chains through to the USA. Impacts and resource use associated with red meat supply are heavily influenced by the production system and less by other components of the supply chain such as transportation or meat processing. Transportation was found to contribute ≤5% of GHG emissions, and water and land
resource use, confirming that food miles is not a suitable indicator of environmental impacts for red meat. Emissions from livestock, soil and fossil fuel consumption were similar in aggregate to other production regions in the world, and LU and dLUC emissions were found to be a moderate emission source for beef, but not lamb, in the current analysis. Deforestation emissions have declined significantly for Australian beef and as a consequence, the emissions attributed to beef from historic clearing events will continue to decline. Reforestation and soil carbon sequestration were shown to have potential to represent a small offset for emissions from beef and lamb production. Water use was found to be relatively low and was drawn predominantly from low water stress catchments. In the first Australian analysis of regional land occupation and consumption of human edible protein inputs, red meat products were found to be produced predominantly from non-arable rangeland areas with small amounts of arable land occupation. Human edible grain inputs were modest for the grass finished systems, displaying a high degree of resource efficiency in the production of high quality food from low quality grass resources, with a low degree of competition with human food sources or other grain users. These results underscore the valuable role in the global food supply chain that ruminants can provide when managed to exploit marginal land which is poorly suited for arable production.

Acknowledgements

This study was funded by Meat and Livestock Australia, project number B.C.CH.2072. Review and input to the project scope from Dr. Tom Davison was appreciated. The contribution of data from the many case study farmers involved in the study was greatly appreciated.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2015.01.073.

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6.3 ADDENDUM

This paper presented results for grass and grain fed beef, and lamb production, and included meat processing and transport to the USA. The results were based on data presented in chapter 3 and chapter 4 (beef) and chapter 5 (lamb).

This paper provided important new information regarding the impacts of meat processing, and also provided results in the context of important export markets rather than production systems or regions.

Inventory data and assumptions associated with meat processing were presented in this paper, and these data were utilised for expanding and revising the results from chapter 3, chapter 4, chapter 5 and chapter 7 (see results in Table 4 and Table 5).

This paper also expanded the analysis to include human edible protein conversion efficiency (HEP-CE) for beef and lamb. The value for grass-fed beef (7.9) was used to determine human edible protein required (HEPR - Figure 3, section 10.5.7), which was the inverse of this value.

The inverse of the HEP-CE value for lamb (2.9) was used to determine HEPR (Figure 3, section 10.5.7). This value was averaged with the HEPR value from merino lambs (see addendum for chapter 7).

Greenhouse gas emissions associated with LU and dLUC were reported in this study for average beef and lamb production systems based on the published, companion study of Henry et al. (2015a). These results were referred to in the discussion, and were included in the estimated total greenhouse gas emissions in Figure 3, section 10.5.7.
7 RESOURCE USE AND GREENHOUSE GAS EMISSIONS FROM THREE WOOL PRODUCTION REGIONS IN AUSTRALIA

7.1 ATTRIBUTION STATEMENT

The paper *Resource use and greenhouse gas emissions from three wool production regions in Australia* was led by S.G Wiedemann, and co-authored by M-J Yan, B.K. Henry and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling flocks and impacts from greenhouse gas emissions
- Development and application of novel methods for determining water use from farm storages
- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and flock outputs, and application of novel allocation methods to handle co-production of wool and liveweight
- Primary data acquisition and data analysis
- Preparation of the manuscript and completion of the peer-review process

Yan contributed to the study and manuscript via:

- Assistance with data collection, collation and analysis
- Assistance with development of figures and tables for the manuscript
- Performing the uncertainty analysis calculations

Henry contributed to the study and manuscript via:

- Assistance with integration of the findings regarding GHG emissions from land use and land use change, which cite an associated publication by Henry, Butler and Wiedemann, and assistance with review and preparation of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with analysis and manuscript preparation.
7.2 INTRODUCTION

This study was the first multi-impact analysis of wool production published in Australia, and followed the beef and lamb studies (chapter 3, chapter 5) using similar methodological approaches. Two companion papers were also written: i) a study of allocation methods was developed as an early output from the project (Wiedemann et al., 2015b) and ii) an analysis of land use and land use change emissions and sequestration opportunities was conducted by the author and colleagues (Henry et al., 2015b).

The study was one of a series of Australian wool LCA papers, including the previous work of Eady et al. (2012) and Brock et al. (2013) which studied GHG emissions from representative farms in WA and southern NSW respectively. The present study built on this knowledge base, applying new allocation methods and expanding the analysis to include multiple impact categories and a wider range of regions.

This study included new insights about the contribution of carbon sequestration to the carbon footprint of wool production, based on field assessments of case study farms. It also expanded the analysis of farm water use to new regions (i.e. Western Australia).

The study provided a robust knowledge base for assessing the environmental impacts from wool production. This has been used subsequently to report the environmental credentials of wool in global supply chains (via the SAC Higg MSI database).
Resource use and greenhouse gas emissions from three wool production regions in Australia

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A B S T R A C T

Australia is the largest supplier of fine apparel wool in the world, produced from diverse sheep production systems. To date, broad scale analyses of the environmental credentials of Australian wool have not used detailed farm-scale data, resulting in a knowledge gap regarding the performance of this product. This study is the first multiple impact life cycle assessment (LCA) investigation of three wool types, produced in three geographically defined regions of Australia: the high rainfall zone located in New South Wales (NSW HRZ) producing super-fine Merino wool, the Western Australian wheat-sheep zone (WA WSZ) producing fine Merino wool, and the southern pastoral zone (SA SPZ) of central South Australia, producing medium Merino wool. Inventory data were collected from both case study farms and regional datasets. Life cycle inventory and impact assessment methods were applied to determine resource use (energy and water use, and land occupation) and GHG emissions, including emissions and removal associated with land use (LU) and direct land use change (dLUC). Land occupation was divided into use of arable and non-arable land resources. A comparison of biophysical allocation and system expansion methods for handling co-production of greasy wool and live weight (for meat) was included.

Based on the regional analysis results, GHG emissions (excluding LU and dLUC) were 20.1 ± 3.1 (WA WSZ, mean ± 2 S.D.) to 21.3 ± 3.4 kg CO2-e/kg wool in the NSW HRZ, with no significant difference between regions or wool type. Accounting for LU and dLUC emissions and removals resulted in either very modest increases in emissions (0.3%) or reduced net emissions by 0–11% depending on pasture management and revegetation activities, though a higher degree of uncertainty was observed in these results. Fossil fuel energy demand ranged from 12.5 ± 4.1 in the SA SPZ to 22.5 ± 6.2 MJ/kg wool (WA WSZ) in response to differences in grazing intensity. Fresh water consumption ranged from 204.3 ± 69.6 L H2O-e/kg wool (NSW HRZ) to 393.7 ± 119.5 L H2O-e/kg wool (NSW HRZ) and followed an opposite trend to water consumption in response to the different levels of water stress across the regions. Non-arable grazing land was found to range from 55% to almost 100% of total land occupation. Different methods for handling co-production of greasy wool and live weight changed estimated total GHG emissions by a factor of three, highlighting the sensitivity to this methodological choice and the significance of meat production in the wool supply chain. The results presented improve the understanding of environmental impacts and resource use in these wool production regions as a basis for more detailed full supply chain analysis.

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1. Introduction

Australia is the largest exporter of greasy wool in the world, trading over 289 thousand tonnes in 2011 (FAO, 2011), from a flock of 68.1 million wool sheep (AWI, 2011), though production has declined in the past two decades (Curtis, 2009). Australian wool production is based on the Merino sheep breed, which produces highly sought-after wool for garment manufacture. Meat
production from lambs and cull-for-age (CFA) breeding animals also represents a valuable co-product.

With increased demand for information regarding the environmental credentials of fibre products from garment manufacturers, retailers, and consumers (Kviseth and Tobiasson, 2011; BSI, 2014; Karim et al., 2014), the need for scientifically-sound whole of supply chain research addressing key environmental impacts and resource use issues is acute. Addressing this need for wool production is more complex than is generally the case for man-made fibres as the latter have relatively consistent and regulated systems for the raw material phase of the supply chain compared to wool.

Life cycle assessment (LCA) is the most widely used tool for reporting the environmental impacts and resource use of products (ISO 2006) and ideally assessment should report on all major environmental impact and resource use categories affected by a product across the full supply chain. A number of sheep studies have focussed on lamb production (Ledgard et al., 2011; Peters et al., 2010a, 2010b; Ripoll-Bosch et al., 2012; Wiedemann et al., 2015c; Williams et al., 2006) though few of these reported impacts for wool. A review by Henry (2011) demonstrated the limitations in data and methodology in past LCA studies, and, to date, only two detailed LCA studies have been published for wool produced in Australia and these reported only the single impact of greenhouse gas (GHG) emissions, excluding land use (LU) and direct land use change (dLU), for cradle to farm-gate wool production, each from a single case study farm (Brock et al., 2013; Eady et al., 2012).

In the absence of detailed studies based on Australian production practices and performance data, the environmental credentials of wool have been modelled using inventory data (i.e. Made-by, 2011) that do not accurately reflect Australian production methods. Given this, and the narrow focus of the case studies to date, the present study aimed to produce a benchmark analysis of water, energy, land and greenhouse gas emissions for three types of Australian Merino wool, produced in three different production systems across the country using a broader farm dataset. Detailed aims are provided in the following section.

2. Materials and methods

2.1. Goal and scope

The study investigated impacts from major Australian wool production regions to provide information to the wool industry, wool fabric users and the general public. The study specifically aimed to i) quantify resource use for energy, water and land, ii) to estimate GHG emissions and removal associated with land use and direct land use change (LU and dLU) from wool production, and iii) to identify impact hotspots in the production system. The system boundary included all supply chain processes associated with the primary production of wool to the farm-gate (Fig. 1). The functional unit was ‘1 kg of greasy wool at the farm gate’.

Impact assessment included global warming using Global Warming Potentials (GWPs) based on the IPCC (Solomon et al., 2007). Fossil fuel energy demand was assessed from an inventory of energy demand throughout the system, and was reported in mega-joules (MJ) with lower heating values (LHV). Stress-weighted water use was assessed using the water stress index (WSI) of Pfister et al. (2009) and reported in water equivalents (H2O-e) after Ridoutt and Pfister (2010). Inventory results were also presented for fresh water consumption and land occupation with methods described in the following sections.

2.1.1. Regions and farming systems

Wool is produced in three broadly defined Australian agro-climatic zones; the high rainfall zone (>600 mm average annual rainfall or a.a.), the wheat-sheep zone (300–600 mm a.a.) and the pastoral zone (<300 mm a.a.) (Hassall & Associates Pty Ltd, 2006). The largest numbers are located in the wheat-sheep and high rainfall zones (~53% and 39%) with smaller numbers in the pastoral zone (Hassall & Associates Pty Ltd, 2006). This study selected farms from geo-spatially defined regions within each zone (see Supplementary material). The defined regions were located in the western wheat-sheep zone (WA WSZ), the eastern high-rainfall zone (northern NSW HRZ) and the southern pastoral zone (central SA SPZ).

The western wheat-sheep region is classified as temperate, with a winter dominant rainfall pattern of 400–550 mm a.a. Within this region, the case study farms were located at an elevation of ~250–300 m above sea level in flat to undulating terrain, near the town of Darken. Temperatures range from an average minimum monthly average of ~6 °C in winter, to a maximum monthly average of ~30 °C in summer. Farms produced wheat and other grains on arable land, and typically grazed sheep on non-arable land, or land being used for pasture leys within the cropping cycle. Grazing is supported by native pastures with introduced clover, predominantly Trifolium subterraneum, and supplied with annual or bi-annual applications of super-phosphate and lime as required. Supplementary feeding and forage crops are used to manage annual feed deficiencies in summer. Wool is produced from large-bodied Merino sheep, producing fine wool (20 µm) and lambs for meat production.

The eastern high rainfall region is a cool temperate environment with a summer dominant rainfall pattern of 700–900 mm a.a. Temperatures range from an average minimum monthly average of ~0 °C in winter, to a maximum average of ~27 °C in summer. Within this region, the case study farms were located at an elevation of ~950–1000 m above sea level in undulating to hilly terrain, near the town of Armidale. Farms are typically mixed grazing enterprises, producing wool, lamb and beef with only small areas of crop land used for forage. Grazing is supported by native pastures with introduced clover, or sown pastures, and is typically supplied with applications of superphosphate every 2–3 years. Small amounts of supplementary feed are used in lower rainfall years and annually during winter. Wool is typically produced from smaller bodied Merino sheep, producing super-fine wool (17 µm) and smaller lambs for meat production.

The southern pastoral region contains large sections of arid (<250 mm) desert lands, with smaller areas of semi-arid (>250 mm, winter dominant) native grasslands or savannas, which support low densities of sheep and cattle, with no cropping and few alternative farming systems available. Supplementary feed is not typically used. Temperatures range from an average minimum monthly average of ~4 °C in winter, to a maximum average of ~34 °C in summer. Within this region, the case study farms were located at an elevation of ~300–350 m in flat to hilly terrain, near the town of Hawker. Because of the low grazing density, the farms studied from this region were very large (>15,000 ha) and management inputs were low. Sheep on the farms studied were typically set-stocked in large paddocks (>2000 ha) and were handled infrequently. Wool is produced from large-bodied Merino sheep, producing medium micron wool (21–22 µm) and lambs for meat production.
2.2. Inventory data

2.2.1. Datasets

Data were collected from 10 case study farms (CSFs) via site visits, interviews and a survey of each farm in 2012–13. An analysis of regional average farms (RAFs) was performed using farm survey data collected from specialist sheep farms as part of the Australian Agricultural and Grazing Industries Survey performed annually by the Australian Bureau of Agricultural and Resource Economics and Sciences. Research methods for this survey are outlined in ABARES (2011). The specialist sheep farm dataset included 34 farms (NSW HRZ, ABARES region 131), 18 farms (WA WSZ, ABARES region 521) and 19 farms (SA SPZ, ABARES region 411) covering five years from 2006 to 2010 (ABARES, 2013). Five years of data were used to account for inter-annual variation as a result of seasonal variation, following recommendations from LEAP (2014).

Land and water resources, and sheep flock characteristics of CSFs and RAFs are presented in Tables 1–4. Sheep numbers, live weights and growth rates were used to model feed intake, manure production and drinking water consumption, and to verify the output of wool and live weight reported. The RAF analysis required additional information to determine the sale weight and age of lambs and sheep leaving the flock. These were determined from reported sale prices ($/lamb) and market average sale prices ($/kg). Replacement ewe numbers were determined from the replacement requirements to maintain flock numbers (i.e. equivalent to annual mortalities and sales of cull breeding sheep) and replacement ewes were assumed to be mated for the first time at 18 months of age. The flocks sold lambs, breeding sheep and older sheep and total live weight sold was an aggregate of all sheep sales. Growth rates were determined from lower-bound estimates of lamb age from the corresponding CSF dataset, resulting in growth rates that were intermediate between the CSFs and values reported by the Commonwealth of Australia (2015).

The inventory of major purchased inputs and land use, along with the major outputs (greasy wool and sheep sales) for the sheep sub-systems are presented in Tables 5 and 6. Transport of livestock and purchased inputs were included. Purchased goods and services (e.g. administration, veterinary services) were modelled based on expenditure, using economic input–output data (Rebitzer et al., 2002). Inventory data were reported in mass units for the CSF dataset. However, the RAF dataset was reported as expenditure and mass of purchased inputs were determined using product prices and disaggregation data supplied in the Supplementary material.

Modelling of energy demand was based on the inventory of purchased goods, services and transport distances (Tables 5 and 6). Capital infrastructure (buildings, fences) and machinery were excluded based on their minor contribution (<1% of impacts) assessed during the scoping phase. Impacts generated off-farm via the use of purchased inputs were modelled using background data were sourced from the Australian life cycle inventory database (Life Cycle Strategies, 2007) where available, or the European Ecoinvent (2.2) database (Swiss Centre for Life Cycle Inventories, 2010). Impacts associated with the use of purchased grains were modelled using feed grain inventories described by the authors (Wiedemann et al., 2010a, 2010b; Wiedemann and McGahan, 2011).

2.2.2. Feed intake and greenhouse gas emissions

In each region, sheep were grazed in open pasture lands year round, with short periods of supplementary feeding in two regions only. Feed intake was modelled using the AFRC (1990) method applied by the Australian National Greenhouse Gas Inventory (NGGI) (Commonwealth of Australia, 2015). The mass and characteristics of supplementary feed (Tables 5 and 6) were collected from farm records, and deducted from modelled feed intake to determine the mass of pasture consumed. Pasture type and pasture characteristics such as crude protein levels were assessed visually during site visits to the CSF. Uncertainty related to the prediction of feed intake for grazing ruminants may be substantial (Popp, 1996), and was accounted for using a range of ±20% for predicted dry matter intake based on the review by Popp (1996).

Livestock greenhouse gas emissions were determined by applying methods outlined in the Australian NGGI (Commonwealth of Australia, 2015) where specific tier two methods were available, or from the IPCC (De Klein et al., 2006). Key factors are provided in the Supplementary material. Uncertainty associated with emission factors was determined from the corresponding IPCC inventory methods (De Klein et al., 2006; Dong et al., 2006).
Methods and inventory data relating to emissions and removals from LU and dLUC were assessed in a parallel study (Henry et al., 2015) which included the impacts of soil carbon change under pastures, and the impact of deforestation and reforestation. Soil carbon and deforestation associated with regional cropping was included using methods outlined in Wiedemann et al. (2015c).

### 2.2.3. Fresh water consumption

Fresh water consumption refers to evaporative losses, or uses that incorporate water into a product that is subsequently not released back into the same river catchment (ISO, 2014). The impact of a change in water yield as a result of dLUC, as recommended by ISO (2014), was assessed using a baseline period of 1990 to make of a change in water yield as a result of dLUC, as recommended by ISO (2014), was assessed using a baseline period of 1990 to make.

\[ I_{\text{water}} = 0.1911 \times t - 2.882 \]

where \( I_{\text{water}} \) = water intake (L/45 kg LW sheep per day); \( t \) = maximum daily air temperature (°C).

The equation is zero when \( t \leq 15 \), when sheep are able to meet their water requirements from pasture intake alone. \( R^2 \) for the equation = 0.84.

Drinking water per sheep accounted for differences in live weight and reproductive status using the method outlined by Luke (1987). Drinking water for cattle was predicted using equations from Ridoutt et al. (2012). All drinking water was modelled as fresh water consumption, because water is lost to the atmosphere via respiration and perspiration, integrated into the product and released outside the river catchment or excreted as urine, which is analogous to irrigation of pasture. Proportions of drinking water supplied from bores, creeks and rivers or farm dams (Table 2) were

### Table 1

<table>
<thead>
<tr>
<th>Parameter</th>
<th>NSW HRZ CSF (n = 3)</th>
<th>WA WSZ CSF (n = 4)</th>
<th>SA SPZ CSF (n = 3)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Annual rainfall (mm)</td>
<td>767</td>
<td>550</td>
<td>264</td>
<td>100 year average from nearest town ( \text{a} ) — SILO climate database (Queensland Government, 2015)</td>
</tr>
<tr>
<td>Average annual evaporation (mm)</td>
<td>1278</td>
<td>1461</td>
<td>2236</td>
<td>101 year average from nearest town ( \text{a} ) — SILO climate database (Queensland Government, 2015)</td>
</tr>
<tr>
<td>Water Stress Index (WSI)</td>
<td>0.011</td>
<td>0.012</td>
<td>0.017</td>
<td>Determined from GIS overlay of Pfister et al. (2009)</td>
</tr>
<tr>
<td>Land resources for the whole farm</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total utilised land area (ha)</td>
<td>878</td>
<td>2820</td>
<td>19,000</td>
<td>Farm data</td>
</tr>
<tr>
<td>Crop land (ha)</td>
<td>0</td>
<td>1294</td>
<td>500</td>
<td>Farm data</td>
</tr>
<tr>
<td>Arable land for pasture (ha)</td>
<td>41</td>
<td>405</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Non arable land (ha)</td>
<td>837</td>
<td>1121</td>
<td>18,500</td>
<td>Farm data</td>
</tr>
<tr>
<td>Sheep flock</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Breeding ewes (no. joined)</td>
<td>2715</td>
<td>5917</td>
<td>2733</td>
<td>Farm data</td>
</tr>
<tr>
<td>Ewe standard reference weight (SRW) (kg/head)</td>
<td>45</td>
<td>55</td>
<td>60</td>
<td>Farm data</td>
</tr>
<tr>
<td>Breeding ewe replacement rate</td>
<td>26</td>
<td>31</td>
<td>33</td>
<td>Farm data</td>
</tr>
<tr>
<td>Breeding ewe mortality rate (%)</td>
<td>2.3</td>
<td>4.3</td>
<td>4.0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Fibre diameter (µm)</td>
<td>17</td>
<td>20</td>
<td>21</td>
<td>Farm data</td>
</tr>
<tr>
<td>Clean wool yield (% greasy)</td>
<td>67</td>
<td>60.8</td>
<td>63</td>
<td>Farm data</td>
</tr>
<tr>
<td>Lambing (% at marking)</td>
<td>86.4</td>
<td>86.5</td>
<td>90</td>
<td>Farm data</td>
</tr>
<tr>
<td>Annual wool clip (total kg greasy)</td>
<td>16,905</td>
<td>45,975</td>
<td>28,277</td>
<td>Farm data</td>
</tr>
<tr>
<td>Annual sheep sales (total kg LW)</td>
<td>97,206</td>
<td>239,899</td>
<td>126,144</td>
<td>Farm data</td>
</tr>
<tr>
<td>Sheep flock characteristics for the case study farms (CSF) based on primary data in the eastern High Rainfall Zone (NSW HRZ), the western Wheat Sheep Zone (WA WSZ) and the Southern Pastoral Zone (SA SPZ).</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

\( a \) NSW HRZ nearest towns include Kentucky, Kentucky South & Dangarsleigh. WA WSZ nearest towns include Darkan, Bokal and Quindanning. SA SPZ nearest towns include Carrietion, Quorn and Hawker.

### Table 2

<table>
<thead>
<tr>
<th>Modelled outputs</th>
<th>NSW HRZ CSF (n = 3)</th>
<th>WA WSZ CSF (n = 4)</th>
<th>SA SPZ CSF (n = 3)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wool sold per breeding ewe (kg greasy/head)</td>
<td>6.2</td>
<td>7.8</td>
<td>10.3</td>
<td>Modelled using annual wool clip and breeding ewe number (Table 1)</td>
</tr>
<tr>
<td>Live weight (LW) sold per breeding ewe (kg LW/head)</td>
<td>35.8</td>
<td>40.5</td>
<td>46.2</td>
<td>Modelled using annual sheep sales and breeding ewe number (Table 1)</td>
</tr>
<tr>
<td>Total flock dry matter intake (t DMI)</td>
<td>1976</td>
<td>4227</td>
<td>2506</td>
<td>Modelled feed intake based on livestock numbers, live weight and metabolizability of the diet using Commonwealth of Australia (2015) method</td>
</tr>
<tr>
<td>Pasture land used for sheep (%)</td>
<td>73.7</td>
<td>100</td>
<td>93.1</td>
<td>Determined from total livestock numbers and modelled feed intake</td>
</tr>
<tr>
<td>Biophysical allocation to wool (%)</td>
<td>35.4</td>
<td>37.8</td>
<td>39.7</td>
<td>Modelled using method outlined in Wiedemann et al. (2015a)</td>
</tr>
<tr>
<td>Farm water model</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Farm dam water supply (%)</td>
<td>77.7</td>
<td>76.0</td>
<td>26.7</td>
<td>Derived from farm water supply system model</td>
</tr>
<tr>
<td>Bore water supply (%)</td>
<td>15.6</td>
<td>10.3</td>
<td>73.3</td>
<td>Derived from farm water supply system model</td>
</tr>
<tr>
<td>Creek water supply (%)</td>
<td>6.7</td>
<td>13.8</td>
<td>0</td>
<td>Derived from farm water supply system model</td>
</tr>
<tr>
<td>Dam density (ML per km²)</td>
<td>8.2</td>
<td>3.3</td>
<td>0.3</td>
<td>Derived from farm water supply system model</td>
</tr>
<tr>
<td>Dam efficiency factor</td>
<td>0.175</td>
<td>0.1</td>
<td>0.075</td>
<td>Derived from farm water supply system model</td>
</tr>
</tbody>
</table>
determined from the survey and site visits for the CSF and verified by an analysis of water supply points using satellite imagery. Losses from the water supply system and dam supply efficiency were modelled using methods outlined in Wiedemann et al. (2015b) which are described briefly here. Where losses associated with the supply of water were caused by the production system, they were attributed to livestock stock. Losses from farm reticulation systems were determined from sources of leakage and evaporation from open tanks and troughs. Evaporation losses from creeks and rivers were endemic to the natural system and were not attributed to livestock. Farm dam water balances were constructed from the inflow, extraction rates, predicted evaporation and seepage using a daily time-step water balance over a 70 year period, using long term rainfall and evaporation data (Jeffrey et al., 2001; Queensland Government, 2015). Catchment runoff (dam inflow) was modelled using USDA-SCS KII curve numbers (USDA NRCS, 2007) with appropriate values determined from site observations of soil type, farming practices and farmer knowledge of the frequency of runoff events. Dam supply efficiency is reported in Wiedemann et al. (2015a).

### Table 3

<table>
<thead>
<tr>
<th>Parameter</th>
<th>NSW HRZ RAF (n = 34)</th>
<th>WA WSZ RAF (n = 18)</th>
<th>SA SPZ RAF (n = 19)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>Annual rainfall (mm)</td>
<td>751</td>
<td>461</td>
<td>243</td>
<td>Long term average from representative towns — Australian Rainman climate database (Clewett et al., 2003)</td>
</tr>
<tr>
<td>Average annual evaporation (mm)</td>
<td>1451</td>
<td>1832</td>
<td>2504</td>
<td>Long term average from representative towns — Australian Rainman climate database (Clewett et al., 2003)</td>
</tr>
<tr>
<td>Water Stress Index (WSI)</td>
<td>0.214</td>
<td>0.012</td>
<td>0.017</td>
<td>Determined from GIS overlay of Pfister et al. (2009)</td>
</tr>
<tr>
<td>Total land area (ha)</td>
<td>929</td>
<td>1804</td>
<td>58,878</td>
<td>Farm data</td>
</tr>
<tr>
<td>Crop land (ha)</td>
<td>0</td>
<td>251</td>
<td>119</td>
<td>Farm data</td>
</tr>
<tr>
<td>Arable land for pasture (ha)</td>
<td>43.4</td>
<td>412</td>
<td>0</td>
<td>Derived from farm data</td>
</tr>
<tr>
<td>Non arable land (ha)</td>
<td>885.6</td>
<td>1141</td>
<td>58,761</td>
<td>Derived from farm data</td>
</tr>
<tr>
<td>Breeding ewes (no. joined)</td>
<td>1516</td>
<td>2179</td>
<td>2885</td>
<td>Farm data</td>
</tr>
<tr>
<td>Ewe standard reference weight (SRW) (kg/head)</td>
<td>50</td>
<td>60</td>
<td>60</td>
<td>Regional average from Commonwealth of Australia (2015)</td>
</tr>
<tr>
<td>Breeding ewe mortality rate (%)</td>
<td>4.0</td>
<td>7.4</td>
<td>8.2</td>
<td>Farm data</td>
</tr>
<tr>
<td>Number of prime lambs sold</td>
<td>339</td>
<td>463</td>
<td>72</td>
<td>Farm data</td>
</tr>
<tr>
<td>Value of prime lambs ($/head)</td>
<td>96</td>
<td>74</td>
<td>62</td>
<td>Farm data</td>
</tr>
<tr>
<td>Total number of lambs sold</td>
<td>618</td>
<td>775</td>
<td>513</td>
<td>Farm data</td>
</tr>
<tr>
<td>Clean wool yield (% greasy)</td>
<td>63.8</td>
<td>61.1</td>
<td>61.6</td>
<td>Farm data</td>
</tr>
<tr>
<td>Lambing (% at marking)</td>
<td>84.6</td>
<td>76.2</td>
<td>69.2</td>
<td>Farm data</td>
</tr>
<tr>
<td>Total wool clip (total kg greasy)</td>
<td>12,454</td>
<td>18,106</td>
<td>28,950</td>
<td>Farm data</td>
</tr>
<tr>
<td>Wool sold per breeding ewe (kg greasy/head)</td>
<td>8.2</td>
<td>8.3</td>
<td>10</td>
<td>Based on annual wool clip and breeding ewe number (Table 3).</td>
</tr>
<tr>
<td>Live weight (LW) sold per breeding ewe (kg LW/head)</td>
<td>34.4</td>
<td>30.1</td>
<td>32.3</td>
<td>Modelled from annual sheep sales and breeding ewe number (Table 3).</td>
</tr>
<tr>
<td>Breeding ewe replacement rate</td>
<td>22</td>
<td>26</td>
<td>28</td>
<td>Replacement rate determined from flock model from adult sheep numbers sold, assuming a static number of breeding ewes maintained in the flock equivalent to the five year flock size average in the dataset</td>
</tr>
<tr>
<td>Annual sheep sales (total kg LW)</td>
<td>52,173</td>
<td>65,677</td>
<td>93,279</td>
<td>Determined from reported number of sheep and lambs sold, sale price of animals (Table 3) and regional sale values to determine mass at sale</td>
</tr>
<tr>
<td>Total flock dry matter intake (t DMI)</td>
<td>1049.4</td>
<td>1264.6</td>
<td>2078.5</td>
<td>Modelled feed intake based on livestock numbers, live weight and metabolizability of the diet using Commonwealth of Australia (2015) method</td>
</tr>
<tr>
<td>Pasture land used for sheep (%)</td>
<td>69.7</td>
<td>88.4</td>
<td>94.4</td>
<td>Determined from total livestock numbers and modelled feed intake</td>
</tr>
<tr>
<td>Biophysical allocation to wool (%)</td>
<td>41.7</td>
<td>46.7</td>
<td>47.2</td>
<td>Modelled using method outlined in Wiedemann et al. (2015a)</td>
</tr>
</tbody>
</table>

### Table 4

<table>
<thead>
<tr>
<th>Description</th>
<th>NSW HRZ RAF (n = 34)</th>
<th>WA WSZ RAF (n = 18)</th>
<th>SA SPZ RAF (n = 19)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Wool sold per breeding ewe (kg greasy/head)</td>
<td>8.2</td>
<td>8.3</td>
<td>10</td>
</tr>
<tr>
<td>Live weight (LW) sold per breeding ewe (kg LW/head)</td>
<td>34.4</td>
<td>30.1</td>
<td>32.3</td>
</tr>
<tr>
<td>Breeding ewe replacement rate</td>
<td>22</td>
<td>26</td>
<td>28</td>
</tr>
<tr>
<td>Annual sheep sales (total kg LW)</td>
<td>52,173</td>
<td>65,677</td>
<td>93,279</td>
</tr>
<tr>
<td>Total flock dry matter intake (t DMI)</td>
<td>1049.4</td>
<td>1264.6</td>
<td>2078.5</td>
</tr>
<tr>
<td>Pasture land used for sheep (%)</td>
<td>69.7</td>
<td>88.4</td>
<td>94.4</td>
</tr>
<tr>
<td>Biophysical allocation to wool (%)</td>
<td>41.7</td>
<td>46.7</td>
<td>47.2</td>
</tr>
</tbody>
</table>
values from Pfister et al. (2009) (Tables 1 and 3). The WSI indicates the portion of fresh water consumption that deprives other users of fresh water, and is thus a measure of scarcity of fresh water. For fresh water consumption in upstream processes of unknown origin, we applied the global average WSI of 0.602 (Ridoutt and Pfister, 2010).

### 2.2.5. Land occupation

Land occupation was determined using a disaggregated land inventory accounting for differences in land type using three categories (measured in m²/yr): i) occupation of non-arable (range-lands) for pasture, ii) occupation of crop land — cultivated for grain or forage crop production, and iii) occupation of arable land for crop production.

### Table 5

Major inputs and outputs for the sheep sub-system on case study farms (CSF) in the eastern High Rainfall Zone (NSW HRZ), the western Wheat Sheep Zone (WA WSZ) and the Southern Pastoral Zone (SA SPZ).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>NSW HRZ CSF (n = 3)</th>
<th>WA WSZ CSF (n = 4)</th>
<th>SA SPZ CSF (n = 3)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Inputs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Land</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>On-farm crop land (ha)</td>
<td>0</td>
<td>241</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Arable land for pasture (ha)</td>
<td>31</td>
<td>405</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Non arable land (ha)</td>
<td>624</td>
<td>1121</td>
<td>17,223</td>
<td>Farm data</td>
</tr>
<tr>
<td>Energy</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Electricity (kWh)</td>
<td>5657</td>
<td>6706</td>
<td>8202</td>
<td>Farm data</td>
</tr>
<tr>
<td>Diesel (L)</td>
<td>2434</td>
<td>9330</td>
<td>6131</td>
<td>Farm data</td>
</tr>
<tr>
<td>Petrol (L)</td>
<td>1866</td>
<td>2655</td>
<td>1750</td>
<td>Farm data</td>
</tr>
<tr>
<td>Fertiliser</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Superphosphate (t)</td>
<td>24</td>
<td>107</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Lime (t)</td>
<td>69</td>
<td>175</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Purchased feed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Protein grains (t)</td>
<td>30</td>
<td>205</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td><strong>Overheads</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Administration ($)</td>
<td>8192</td>
<td>20,850</td>
<td>6964</td>
<td>Farm data</td>
</tr>
<tr>
<td>Veterinary products ($)</td>
<td>15,478</td>
<td>28,156</td>
<td>7620</td>
<td>Farm data</td>
</tr>
<tr>
<td>Herbicides ($)</td>
<td>807</td>
<td>0</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Transport (t km)</td>
<td>4901</td>
<td>39,308</td>
<td>16,039</td>
<td>Farm data</td>
</tr>
<tr>
<td><strong>Outputs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Greasy wool (kg)</td>
<td>16,905</td>
<td>45,975</td>
<td>28,277</td>
<td>Farm data</td>
</tr>
<tr>
<td>Sheep sales (kg LW)</td>
<td>97,206</td>
<td>239,899</td>
<td>126,144</td>
<td>Farm data</td>
</tr>
</tbody>
</table>

* Farm data reported expenditure. Mass of purchased inputs for the sheep sub-system determined using methods outlined in the Supplementary material.

### Table 6

Major inputs and outputs for the sheep sub-system on regional average farms (RAF) in the eastern High Rainfall Zone (NSW HRZ), the Western Wheat Sheep Zone (WA WSZ) and the southern Pastoral Zone (SA SPZ).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>NSW HRZ RAF (n = 34)</th>
<th>WA WSZ RAF (n = 34)</th>
<th>SA SPZ RAF (n = 19)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Inputs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Land</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>On-farm crop land (ha)</td>
<td>0</td>
<td>74</td>
<td>7</td>
<td>Farm data</td>
</tr>
<tr>
<td>Arable land for pasture (ha)</td>
<td>30</td>
<td>363</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Non arable land (ha)</td>
<td>617</td>
<td>1004</td>
<td>55,234</td>
<td>Farm data</td>
</tr>
<tr>
<td>Energy</td>
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<td></td>
</tr>
<tr>
<td>Electricity (kWh)</td>
<td>7162</td>
<td>3769</td>
<td>6998</td>
<td>Farm data</td>
</tr>
<tr>
<td>Diesel (L)</td>
<td>2747</td>
<td>5714</td>
<td>10,753</td>
<td>Farm data</td>
</tr>
<tr>
<td>Petrol (L)</td>
<td>2106</td>
<td>2218</td>
<td>3069</td>
<td>Farm data</td>
</tr>
<tr>
<td>Fertiliser</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Superphosphate (t)</td>
<td>19</td>
<td>62</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Lime (t)</td>
<td>3</td>
<td>44</td>
<td>0</td>
<td>Farm data</td>
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<tr>
<td>Purchased feed</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Protein grains (t)</td>
<td>36</td>
<td>54</td>
<td>5</td>
<td>Farm data</td>
</tr>
<tr>
<td><strong>Overheads</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Administration ($)</td>
<td>4579</td>
<td>6238</td>
<td>6699</td>
<td>Farm data</td>
</tr>
<tr>
<td>Veterinary products ($)</td>
<td>8651</td>
<td>8424</td>
<td>6822</td>
<td>Farm data</td>
</tr>
<tr>
<td>Herbicides ($)</td>
<td>391</td>
<td>2926</td>
<td>0</td>
<td>Farm data</td>
</tr>
<tr>
<td>Transport (t km)</td>
<td>2649</td>
<td>12,543</td>
<td>7616</td>
<td>Farm data</td>
</tr>
<tr>
<td><strong>Outputs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Greasy wool (kg)</td>
<td>12,454</td>
<td>18,106</td>
<td>28,950</td>
<td>Farm data (see description under Table 3)</td>
</tr>
<tr>
<td>Sheep sales (kg LW)</td>
<td>52,173</td>
<td>65,677</td>
<td>93,279</td>
<td>Determined from reported number of sheep and lambs sold, sale price of animals (Table 3) and regional sale values to determine mass at sale</td>
</tr>
</tbody>
</table>

* Farm data reported expenditure. Mass of purchased inputs for the sheep sub-system determined using methods outlined in the Supplementary material.
pasture. The proportion of land in each category was determined from information provided by the farmers, field observations and analysis of satellite imagery for the CSF. Total land occupation and crop land occupation was reported in the ABARES dataset and was used for the RAF analysis. Non-crop land was determined from the difference between total land area and reported crop land. In this remaining area, we determined the relative proportions of non-farming land, arable pasture and rangeland from equivalent proportions in the CSF dataset for each region.

2.3. Handling co-production

A number of co-products were produced from the farm systems. Sheep farms typically also produced other livestock and grain, which was handled by dividing the sub-systems and accounting for each separately (see Fig. 1).

In most cases, inputs could be divided because they were specific to one system only. Livestock systems were divided based on relative feed requirements, which was causally related to land occupation and to stocking density. The proportion of grazing land used for sheep is reported in Tables 2 and 4. On the case study farms, inputs associated with the cropping system were separated by the farmers. Further detail of the methods applied to separate cropping systems in the RAF dataset is provided in the Supplementary material. Whole farm inputs (overheads, such as electricity use) remaining after the system separation processes were a minor contribution to total impacts, and were divided on the basis of land occupation which aligned to the biological separation process applied for grazing livestock. Interactions between sheep and grain production included the grazing of residuals after crop harvest, benefiting the sheep system, and weed control which benefitted the crop system. The primary benefit from the crop system to sheep was from the consumption of grain spilled on the ground after harvest, and weeds growing in the stubble, rather than crop residues per se (Butler and Croker, 2006). Considering that spilled grain is a waste product from the cropping cycle and grazing weeds is mutually beneficial to both systems, the net contribution of cropping to the sheep system from stubble grazing was considered negligible and no impacts from the cropping cycle were attributed to sheep or vice versa.

Handling co-production of wool and live weight (for meat) was modelled following Wiedemann et al. (2015a) using the protein mass allocation (PMA) method, with a system expansion process used for comparison. The protein mass of greasy wool was estimated by multiplying greasy wool mass by clean wool content (Tables 1 and 3) and assuming a dry matter content of clean wool of 84% and a 100% protein content for dry, clean wool. The protein content of live weight was assumed to be 18% (Wiedemann and Yan, 2014) based on body composition. As a comparison, a system expansion (SE) approach was applied using two scenarios where live weight from the Merino sheep system resulted in avoided live weight production from either an alternative meat sheep flock or from beef cattle production, after Wiedemann et al. (2015a). To account for the lower carcass yield of Merinos compared with meat sheep, a factor of 0.95 was used so that 100 kg of Merino LW was considered equivalent to 95 kg of LW from the avoided meat sheep flock, based on MLA (2003). Two combinations of alternative meat sheep breeds were explored. A composite, crossbreeding system based on Border Leicester crossbred ewes and Poll Dorset rams was chosen for NSW HRZ and WA WSZ systems, while Dorper breed sheep that are well suited to pastoral zone conditions was chosen for SA SPZ system. Dorper sheep produce a very small amount of wool and shed their fleece naturally each year, thereby producing no saleable wool. The Border Leicester crossbreeding system produces wool for interior textiles rather than garment manufacture. In order to use the crossbreeding system to substitute for meat from Merinos, a second substitution process was required to take into account the change in production of interior textiles wool, where the change in this wool product was substituted for nylon at a 1:1 ratio, using nylon processes from Ecoinvent. Inventory data for modelling the alternative beef production systems were collected from the farms that also produced beef, and were augmented with regional data from the ABARES survey to ensure productivity levels were typical of the regions. When substituting beef with sheep meat from the Merino flocks, an equivalence factor of 0.85 was applied to account for differences in carcass yield (Wiedemann and Yan, 2014). The final results of system expansion were averaged across the two live weight substitution scenarios.

2.4. Analysis

Modelling was conducted using SimaPro 8.0 (Pré-Consultants, 2014). Two types of uncertainties in the input variables were considered: alpha and beta uncertainties, after Leinonen et al. (2012). Alpha uncertainty describes the variations among farms reflecting the primary datasets. Beta uncertainty describes the uncertainties in the model and took into account uncertainty in the prediction of feed intake, application of GHG emission factors (see Supplementary material) and uncertainty in background processes based on the applied datasets (see Supplementary material). Alpha and Beta uncertainty was assessed using a Monte Carlo analysis in SimaPro 8.0 (Pré-Consultants, 2014), using one thousand iterations to provide a 95% confidence interval for results. Results were presented using the mean ± 2S.D., and both alpha and beta uncertainties were used to calculate the S.D. As beta uncertainty was shared by all systems, comparison of the mean results between regions was based on alpha uncertainties only, and significant differences were determined using the following equation of Wiltshire et al. (2009):

$$z = \frac{100\sqrt{\frac{A}{CV_A^2} + \frac{B}{CV_B^2}}}{A + B}$$

where $A$, $B$ are the mean values and $CV_A$ and $CV_B$ are coefficients of variance of the two systems compared. In addition, multiple linear regression (MLR) was conducted in R (R Development Core Team, 2014) to determine the factors that most influenced GHG emissions.

3. Results

Excluding LU and dLUC, GHG emissions from wool production varied from 19.5 ± 4.1 kg CO$_2$-e/kg wool (mean ± 2S.D.) in the SA SPZ CSF, to 25.1 ± 4.8 kg CO$_2$-e/kg wool in the NSW HRZ CSF. GHG emissions were dominated by enteric methane, which contributed from 79 to 86% in the RAF dataset, and up to and 89% in the CSF dataset. Nitrous oxide emissions, mainly from animal manure, ranged from 10 to 11% in the RAF dataset and from 9% (SA SPZ and NSW HRZ) to 11% (WA WSZ) in the case study dataset. Carbon dioxide emissions contributed between 4% (SA SPZ) and 9% (WA WSZ) in the RAF dataset and from 2% (SA SPZ) to 7% (WA WSZ) in CSF dataset. These emissions were primarily associated with fossil energy demand, though in the WA WSZ, the elevated CO$_2$ emissions were also partially in response to lime application. Linear regression of wool production per breeding ewe and GHG emissions showed this indicator explained 0.79 of variability (see Fig. 3):
\[ Y = 30.23 - 1.04X \quad (R^2 = 0.79) \]

where \( X \) is flock wool production per breeding ewe (total flock wool production divided by ewes joined); \( Y \) is GHG emissions (kg CO\(_2\)-e/kg greasy wool).

For crop land, GHG emissions from LU and dLUC varied from 0.1 to 0.4 kg CO\(_2\)-e/kg greasy wool in the RAF across the three regions (Table 7). GHG removals (indicated by negative emission values) from LU from fertilised pastures varied from −0.8 to 0.0 kg CO\(_2\)-e/kg greasy wool for the RAF analysis between regions, depending on assumptions regarding soil carbon sequestration under pasture. Corresponding GHG removals from vegetation regrowth of planted trees and shrubs varied from −1.6 to 0.0 kg CO\(_2\)-e/kg greasy wool in the RAF dataset, depending on region (see Table 7). In the CSF dataset, removals associated with pasture ranged from −1.1 to 0.0 kg CO\(_2\)-e/kg greasy wool and removals associated with vegetation regrowth of planted trees and shrubs varied from −2.4 to −0.3 kg CO\(_2\)-e/kg greasy wool.

Fossil fuel energy demand ranged from 11.2 ± 7.4 (NSW HRZ CSF) to 22.5 ± 6.2 MJ/kg wool (WA WSZ) in the RAF dataset with impacts being significantly higher in the WA WSZ (Fig. 2). Farm energy demand (fuel and electricity) was the largest contributor (28–83%) across all regions. In the NSW HRZ and WA WSZ the next largest contributors were fertiliser/pesticides (12–36%) and animal health services (8–23%). Energy demand followed a similar trend in the CSF dataset for the NSW HRZ and WA WSZ, though results were significantly lower for the SA SPZ CSF when compared against the other states and compared to the SA SPZ RAF analysis.

Fresh water consumption ranged from 204.3 ± 59.1 in the NSW HRZ RAF to 393.8 ± 123.8 L/kg greasy wool in the WA WSZ RAF (Table 8). Fresh water consumption was dominated by losses from the farm water supply system across all regions (77–85%), followed by livestock drinking water (13–22%). Evaporative losses from farm dams were the largest contributor to farm water supply losses. Stress weighted water use was significantly lower than fresh water consumption for all regions, ranging from 6.2 ± 3.0 (NSW HRZ CSF) to 74.6 ± 119.5 L H\(_2\)O-e/kg greasy wool in the NSW HRZ RAF.

The land occupation assessment showed a distinct variation between regions in crop land occupation, ranging from 0.03 ± 0.02 in the SA SPZ CSF, to 52.9 ± 15.2 m\(^2\)/kg greasy wool in the WA WSZ (RAF) wheat shee zone. Arable pasture land occupation ranged from negligible levels in the SA SPZ to 93.6 ± 22.3 m\(^2\)/kg greasy wool in the WA WSZ RAF, while non-arable pasture land occupation ranged from 92.1 ± 30.0 (WA WSZ CSF) to 9005.4 ± 3150.9 m\(^2\)/kg greasy wool in the SA SPZ RAF (Fig. 2). Pasture land occupation was lower in the CSF dataset for each region, as a result of higher stocking rates compared to the regional average.

### 4. Discussion

Australia has three major wool producing regions which vary substantially in terms of rainfall, land area and land type, management systems and sheep type. The study considered three representative areas within each region, using two datasets. Despite the very large differences in sheep type and biophysical resources, differences in impacts were relatively small for some impacts such as greenhouse gas emissions intensity. Application of two datasets improved the representativeness and specificity of results. We found the CSF dataset to provide highly detailed data regarding flock management and biophysical resources such as land and water, albeit for a limited number of farms in each region and for one or two years only. In contrast, the RAF dataset provided a larger number of farms in each region, and repeated measures over a longer time frame (five years), but contained less detail regarding some biophysical resources and flock management. This approach provided an internal comparison within each region and improved the overall representativeness of the regional results. To improve the transparency of the results, impacts for sheep meat determined using biophysical allocation are also presented in the Supplementary material.

#### 4.1. Greenhouse gas emissions and removals

GHG emissions (excluding LU and dLUC) were not significantly different between regions. However, a regression analysis of individual farms in the CSF dataset revealed a trend towards higher impacts from systems where wool yield per sheep was lower (i.e. NSW HRZ CSF). Differences in wool and meat production per ewe were largely associated with the strain of Merino sheep bred in each region. Superfine Merinos (from the high rainfall zone in the present study) typically have lower body weights and produce less wool per head than fine and medium wool Merinos. In both the WA WSZ and SA SPZ sheep systems, there was a greater emphasis on breeding for lamb production than in the NSW HRZ farms analysed, which corresponded to a greater mass of live weight produced per breeding ewe in these regions. These sheep systems are also more common in the lower rainfall climates in these regions. Interestingly, the very large differences in production intensity and land resources did not correspond to major differences in emissions, because productivity per breeding ewe was maintained by using lower stocking rates and different strains of Merinos in the lower rainfall areas. Emissions intensity was similar to Eady et al. (2012) and Brock et al. (2013) when differences in allocation procedure and production intensity were taken into account (Fig. 3). The research farm system studied by Brock et al. (2013) had wool production of 13.2 kg greasy wool per ewe joined; significantly

### Table 7

<table>
<thead>
<tr>
<th>Emissions and removals</th>
<th>NSW HRZ RAF</th>
<th>WA WSZ RAF</th>
<th>SA SPZ RAF</th>
<th>NSW HRZ CSF</th>
<th>WA WSZ CSF</th>
<th>SA SPZ CSF</th>
</tr>
</thead>
<tbody>
<tr>
<td>kg CO(_2)-e/kg greasy wool</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Soil carbon — crop land</td>
<td>0.4</td>
<td>0.3</td>
<td>0.1</td>
<td>0.2</td>
<td>0.2</td>
<td>0.0</td>
</tr>
<tr>
<td>Soil carbon — pasture</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower estimate</td>
<td>−0.8</td>
<td>0.0</td>
<td>0.0</td>
<td>−1.1</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Upper estimate</td>
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<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Vegetation</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower estimate</td>
<td>−1.6</td>
<td>−0.4</td>
<td>0.0</td>
<td>−1.6</td>
<td>−0.4</td>
<td>−2.4(\text{a})</td>
</tr>
<tr>
<td>Upper estimate</td>
<td>−1.1</td>
<td>−0.3</td>
<td>0.0</td>
<td>−1.1</td>
<td>−0.3</td>
<td>−2.4(\text{a})</td>
</tr>
<tr>
<td>Total LU and dLUC emissions or removals</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Lower estimate</td>
<td>−1.9</td>
<td>−0.1</td>
<td>0.1</td>
<td>−2.5</td>
<td>−0.1</td>
<td>−2.4</td>
</tr>
<tr>
<td>Upper estimate</td>
<td>−0.7</td>
<td>0.0</td>
<td>0.1</td>
<td>−0.8</td>
<td>0.0</td>
<td>−2.4</td>
</tr>
</tbody>
</table>

\(\text{a}\) Only one estimate provided.
higher than the NSW HRZ CSF (6.2 kg) and NSW HRZ RAF (8.2 kg) assessed here, suggesting that much higher wool production is possible in this region, potentially leading to lower GHG impacts. Emissions were also similar to the supplementary results presented for wool by Wiedemann et al. (2015d) who studied Australian cross-bred sheep systems focussed on lamb production.

Wool farms generate both emissions and removals of greenhouse gases, though the latter have not previously been considered in wool LCAs. While emissions from livestock and energy sources can be modelled using well-defined methods, the determination and attribution of emissions and removals from LU and dLUC sources is more complex and uncertain, particularly at the regional scale. In a parallel study by the authors, Henry et al. (2015) estimated CO2 removals in planted exotic pines and mixed native species of 4.4 and 2.0 t CO2 per ha per year, respectively for the same NSW HRZ and WA WSZ regions, and sequestration of 0.07 t CO2 per ha per year over 100 years for chenopod shrub lands of the SA SPZ CSF. Sequestration of soil organic carbon in improved permanent pastures in the NSW HRZ was evaluated to be highly uncertain and small but potentially significant over large areas of pasture land (Henry et al., 2015).

4.2. Water use

This study presents the first wool specific analysis of water use with comprehensive LCA methods to the authors’ knowledge. Water use was dominated by supply losses and to a lesser extent direct drinking water requirements, and no farms used irrigation water for pasture production. Water losses were highest where the reliance on water from small farm dams was high and evaporation losses were also high. For regions with very high annual evaporative losses such as the SA SPZ, even moderate reliance on dams (27% of supply) resulted in large losses. Dam efficiency was primarily influenced by net evaporation, dam density (total volume stored per km²) and surface area to volume ratio. Dam densities were within the range reported by Nathan and Lowe (2012) but modelled extractions for livestock drinking water as a proportion of dam volume were much lower in the present study (see Table 2) than the assumptions made by these authors. Supplementary data from Wiedemann et al. (2015d) showed that water use from wool in cross-bred sheep systems focussed on lamb production could be higher (up to 741.4 L/kg greasy wool) where irrigation is used. Stress weighted water use results showed much lower values than fresh water consumption. The exception was the NSW HRZ RAF, which had an average WSI of 0.214, driven mainly by an area of higher water stress located in the southern part of this region. This finding is important for a globally traded product such as wool; the impact of using water to produce wool in these Australian regions is comparatively low both in terms of competitive water uses (i.e. for human consumption or industry) or the environment.

4.3. Fossil energy demand

Fossil energy demand varied significantly in response to production intensity, with the highest values observed in the WA WSZ where fertiliser and pesticide inputs associated with pasture and forage were much higher. In contrast, fertiliser was lower in the extensive pastures.

Table 8

<table>
<thead>
<tr>
<th></th>
<th>NSW HRZ RAF</th>
<th>WA WSZ RAF</th>
<th>SA SPZ RAF</th>
<th>NSW HRZ CSF</th>
<th>WA WSZ CSF</th>
<th>SA SPZ CSF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total fresh water consumption (L)</td>
<td>204.3 a</td>
<td>393.8 b</td>
<td>379.7 b</td>
<td>238.7 a</td>
<td>359.7 b</td>
<td>322.4 a b</td>
</tr>
<tr>
<td>Livestock drinking (L)</td>
<td>43.3</td>
<td>50.4</td>
<td>48.7</td>
<td>51.0</td>
<td>45.9</td>
<td>72.0</td>
</tr>
<tr>
<td>Drinking water supply losses (L)</td>
<td>156.9</td>
<td>335.0</td>
<td>294.5</td>
<td>184.9</td>
<td>304.9</td>
<td>250.3</td>
</tr>
<tr>
<td>Other minor inputs (L)</td>
<td>4.0</td>
<td>8.3</td>
<td>0.4</td>
<td>2.7</td>
<td>8.9</td>
<td>0.1</td>
</tr>
<tr>
<td>Stress weighted water use (L H2O-e)</td>
<td>74.6 d</td>
<td>215.5</td>
<td>11 b</td>
<td>6.2 a</td>
<td>13.4 b</td>
<td>9.2 a b</td>
</tr>
</tbody>
</table>

Different letters indicate significant differences between cases based on alpha uncertainty.
management systems used in the SA SPZ, resulting in lower energy demand. Few other studies were found reporting energy demand for wool, though the results presented here were of a similar order to the 13.4 MJ/kg greasy wool for one study in New Zealand (Barber and Pellow, 2006) and tended to be slightly higher than wool from cross-bred sheep systems (Wiedemann et al., 2015d).

4.4. Land occupation

Land occupation for wool production varied more than any other factor in the present study, in response to differences in the underlying land resources available for sheep production across the regions and to differences in the level of supplementary feed used. Crop land occupation was low in the NSW HRZ and the SA SPZ because farms in these regions had less land available for grain and forage cropping compared to the WA WSZ, and utilised only small amounts of purchased supplementary feed. Arable land resources represent only ~4% of national land mass (Lesslie and Mewett, 2013) making this the most constrained land resource in Australia. In contrast, occupation of non-arable land may not result in as high a degree of modification to the natural ecosystem and such land is less suitable for alternative agricultural production, being generally limited to grazing by ruminants. In contrast, crop production results in a higher degree of disturbance of soil and vegetation than grazing. Considering the importance of land type for understanding competitive resource use and environmental impact, we consider a simple analysis of forage cropping compared to the WA WSZ, and utilised only small amounts of purchased supplementary feed. Arable land resources represent only ~4% of national land mass (Lesslie and Mewett, 2013) making this the most constrained land resource in Australia. In contrast, occupation of non-arable land may not result in as high a degree of modification to the natural ecosystem and such land is less suitable for alternative agricultural production, being generally limited to grazing by ruminants. In contrast, crop production results in a higher degree of disturbance of soil and vegetation than grazing. Considering the importance of land type for understanding competitive resource use and environmental impact, we consider a simple analysis of forage cropping compared to the WA WSZ, and utilised only small amounts of purchased supplementary feed. Arable land resources represent only ~4% of national land mass (Lesslie and Mewett, 2013) making this the most constrained land resource in Australia.

4.5. Sensitivity analysis

Allocating impacts on the basis of protein mass resulted in a high allocation of impacts to wool compared to live weight (impacts presented in the Supplementary material) because of the high protein density of wool. Protein mass was determined from product mass and estimated protein content of greasy wool and the live weight of animals sold. To test the sensitivity of the model to assumptions relating to protein content, we varied the wool yield factor within the highest known monthly yield variance measured by the Australian Wool Testing Authority (AWTA, 2015), which was 5.5 percentage points for the WA WSZ region. Variance in this factor resulted in a maximum 5% change in impacts between the highest and lowest values. Similarly, a change in live weight protein yield of one percentage point (from 18% to 19%) reduced impacts by 3%.

In the present study we also applied a system expansion approach for comparison to the selected allocation method (Wiedemann et al., 2015a). System expansion results (see Fig. 4) are presented as a proportion of the biophysical results to demonstrate the effect of changing methods. On average, impacts were 70% lower for GHG, while stress weighted water use was much lower and was negative. These lower impacts resulted from the relatively lower efficiency of the expanded sheep and beef systems. Using these assumptions, we found that changes in wool production may have significantly less impact on total GHG and water use than would be suggested from the benchmarking results because of changes in the meat supply system. Energy demand was 65% higher using SE, due to the energy requirements for alternative fabric (nylon) production to replace wool in the avoided meat sheep scenario. Sensitivity to changes in the co-product system has also resulted in substantially different results between allocation and system expansion in the dairy sector (Cederberg and Stadig, 2003; Flysjø et al., 2011; Zehetmeier et al., 2012). Considering the sensitivity of the results to this methodological aspect, further analysis using consequential modelling is expected to be important for understanding impacts from changing wool supply and demand or investigating mitigations that result in changed production. Further to this, benchmarking results determined using allocation may not provide an accurate picture of the change in environmental impact resulting from a change in supply and demand, because the induced change in the co-product system has not been taken into account.

We also tested the model to sensitivity in assumptions regarding lamb sale age in the RAF model, which was a variable

![Fig. 3. Regression of wool production per breeding ewe and greenhouse gas emissions.](image-url)
determined from sale values and typical sale age for the markets available in each region. We modelled scenarios where the sale age of lambs was reduced by 3 months (i.e. from 12 months to 9 months) at a fixed sale weight by increasing growth rate. This was found to result in modest changes (~3%) to impacts for wool. Regarding GHG modelling assumptions, we found a modest reduction (~4%) in GHG emissions when IPCC (Dong et al., 2006) enteric methane assumptions were applied. While not tested here, the sensitivity of impacts to alternative feed intake prediction methods has been highlighted by Brock et al. (2013) and may result in up to 20% difference in GHG impacts for wool depending on what method is applied. The method used here is applied in the Australian GHG inventory and is considered appropriate for the scale of the assessment. In the RAF model, water supply ratios at the regional level were assumed to follow the ratios determined from the smaller CSF dataset. We found that a 10% increase in the water supplied from farm dams rather than bores or rivers increased fresh water consumption by up to 30% for the SA SPZ, where water supply efficiency from dams was lowest compared to other water sources. Changes were less pronounced (10%) in the NSW HRZ because of the comparatively better dam supply efficiency. The sensitivity of the regional model to this factor suggests further research is warranted to improve the regional assessment of farm water supply. Within the fossil fuel energy assessment, the system separation method was also a sensitive assumption because energy was influenced to a greater extent by farm overheads than other impact categories.

5. Conclusions

This study addresses the lack of farm-level production information regarding the environmental impacts and resource use associated with producing Australian wool by presenting results for three significant regional production zones. While not representative of the whole country, the study significantly expands the knowledge base regarding Australian wool production to the farm gate. The study showed significant differences in some impacts from region to region, influenced by production intensity, the level of inputs and climate. Arable land occupation and energy demand was highest in the mixed grazing and cropping regions where larger amounts of supplementary feed grown on arable land was used for sheep production. The results also showed that non-arable land comprises the largest proportion of total land occupation, which indicated low resource use for crop land that can be used for other fibre and food production systems. Water resource use was highest in production regions with low annual rainfall and high evaporation. Applying the appropriate WSI showed wool to have a relatively low impact on constrained water resources in the three regions, with an exception made for the NSW HRZ RAF. Wool production per breeding ewe explained a high proportion of the variability in GHG emissions intensity (excl. LU and dLuc), highlighting the importance of production efficiency as a means to reduce emissions. Though more uncertain, inclusion of LU and dLuc resulted in lower net emissions, or very modest increases in emissions, than if these emissions were excluded. Application of an alternative system expansion method for handling co-production substantially changed results, highlighting the sensitivity of these results to changes in the co-product system. Thus, further research is required using consequential analysis methods to more accurately determine the environmental impacts from a change in wool production.

Acknowledgements

The authors thank Australian Wool Innovation for financially supporting this project, the farm managers who supplied data and the extension officers who assisted with selection of case study farms.

Appendix A. Supplementary material

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jclepro.2016.02.025.

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7.3 ADDENDUM

This paper focused on wool production and presented results for sheep meat in the supplementary material. For convenience, these results are repeated here in Table 3.

Table 3. Environmental impacts of 1 kg of merino sheep liveweight from case study farms (CSF) and regional average farms (RAF) in High Rainfall Zone (HRZ), the Western Wheat Sheep Zone (WSZ) and the southern Pastoral Zone (SPZ) using biophysical allocation

<table>
<thead>
<tr>
<th></th>
<th>HRZ CSF</th>
<th>HRZ RAF</th>
<th>WSZ CSF</th>
<th>WSZ RAF</th>
<th>SPZ CSF</th>
<th>SPZ RAF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Global Warming (kg CO₂-e / kg LW)</td>
<td>8.9</td>
<td>8.9</td>
<td>8.2</td>
<td>9.4</td>
<td>7.7</td>
<td>9.5</td>
</tr>
<tr>
<td>Fossil energy (MJ / kg LW)</td>
<td>3.9</td>
<td>7.0</td>
<td>6.5</td>
<td>10.5</td>
<td>2.9</td>
<td>5.9</td>
</tr>
<tr>
<td>Fresh water consumption (L / kg LW)</td>
<td>84.5</td>
<td>83.6</td>
<td>135.9</td>
<td>185.7</td>
<td>128.0</td>
<td>179.2</td>
</tr>
<tr>
<td>Stress weighted water use (L H₂O-e / kg LW)</td>
<td>2.2</td>
<td>31.1</td>
<td>5.0</td>
<td>10.0</td>
<td>3.6</td>
<td>5.2</td>
</tr>
</tbody>
</table>

The sheep meat results were modified to enable consistent reporting with other lamb and meat production systems by expanding the system boundary to include meat processing, and revising the reference flow/functional unit from ‘kilograms of liveweight’ to ‘kilograms of boneless meat’ (see results in Table 4 and Table 5).

Meat processing impacts were modelled using inventory data presented in chapter 6 (Table 3) and meat processing assumptions relating to carcase yield were presented in section 2.2.2 of chapter 6. Retail lamb (bone-in) was converted to boneless by assuming a meat yield of 76% after Wiedemann & Yan (2014).

The HEPR was determined by calculating the average percentage of human edible protein in the diet of the RAF sheep flocks, allocated to one kilogram of boneless meat produced. Human edible protein was associated with supplementary grain use, which was reported in the study. Human edible protein was calculated from the protein fraction of each type of supplementary grain and summed across the supply chain to report results relative to boneless lamb.
Total kilograms of human edible protein consumed by the flock and allocated to meat was 0.2 kg, per kilogram of boneless lamb. The protein percentage of boneless lamb was assumed to be 19% and HEPR was 1.0. Average HEPR reported in Figure 3, section 10.5.7, was an average of the results shown here, and the results reported for lamb in chapter 6. The average GHG emissions reported in Figure 3, section 10.5.7 were calculated from the average RAF merino systems and the average RAF lamb results, reported in chapter 6 (after transport impacts to the USA were removed).
8 RESOURCE USE AND ENVIRONMENTAL IMPACTS FROM
AUSTRALIAN CHICKEN MEAT PRODUCTION

8.1 ATTRIBUTION STATEMENT

The paper *Resource use and environmental impacts from Australian chicken meat production* was led by S.G Wiedemann and co-authored by E.J McGahan and C M Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
- Development of the manure management system model
- Development and application of grain processes used in the modelling of feed inputs
- Data acquisition and data analysis, including impact assessment
- Modelling of grow out units and meat processing
- Allocation methodology development
- Preparation of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model

Murphy contributed to the study and manuscript via:

- Assistance with graphs and manuscript preparation
8.2 INTRODUCTION

The chicken meat paper was the final output from a series of chicken meat LCA, benchmarking and GHG emissions projects, initiated in 2010. The initial research was used to identify sensitive GHG emission factors and guide research to quantify manure emissions from chicken meat, which was subsequently published (Wiedemann et al., 2016d). The findings from this research was incorporated in the published LCA study to improve the rigour of the GHG emission assessment.

The study included a detailed analysis of impacts from the meat chicken supply chain, including breeding, grow-out and meat processing. The initial research was used to identify sensitive GHG emission factors and guide research to quantify manure emissions from chicken meat. This guided the first Australian study that measured manure excretion and GHG emissions from chicken meat production (Wiedemann et al., 2016d). The findings from this research were incorporated in the published LCA study, providing a rigorous the GHG emission assessment.

The study provides one of the only analyses of energy, water and land occupation for chicken meat in the global literature and provides a robust analysis for Australian chicken meat production.
Resource use and environmental impacts from Australian chicken meat production

S.G. Wiedemann*, E.J. McGahan, C.M. Murphy

Integrity Ag Services, Highfields, Queensland, Australia

**A B S T R A C T**

Agri-food industries such as chicken meat production face increasing pressure to quantify and improve their environmental performance over time, while simultaneously increasing production to meet global demand. Using life cycle assessment, this study aimed to quantify resource use, environmental impacts and hotspots for Australian chicken meat production using updated inventories and new methods. Two contrasting states; Queensland, and South Australia, and two housing systems; conventional and free range were analysed to indicate the variation expected between regions and systems. Lower impacts were observed per kilogram of chicken meat produced in South Australia compared to Queensland for fossil fuel energy, greenhouse gas (including land use and direct land use change) and fresh water consumption (18.1 and 21.4 MJ, 2.8 and 3.4 kg CO2-e, 38 and 111 L respectively), but not arable land or stress weighted water use (22.5 and 14 m², 36 and 26 L H2O-e respectively). Feed production was the largest contributor to all impact categories, and also showed the largest variation between regions, highlighting the importance of spatially specific feed grain datasets to determine resource use and greenhouse gas from chicken meat production. While the feed conversion ratio was lower in South Australia, this was found to be less significant than differences related to crop yield, irrigation water use and the use of imported feed ingredients, suggesting that incremental improvements in feed conversion ratio will result in lower impacts only when feed inputs and production systems do not change. Fresh water consumption was lower in South Australia, but did not correlate with stress weighted water use (lower in Queensland), highlighting that volumetric water use is not a reliable indicator of the impact of water use. We did not observe substantial differences between conventional and free range production when feed related differences were removed, because key productivity factors such as feed conversion ratio were similar between the two housing types in Australia. While results were found to vary between regions, total greenhouse gas emissions were low from these Australian supply chains, and resource use was moderate. Expansion of the study to include additional regions and impact categories is recommended in future benchmarking studies.

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1. Introduction

Food production supply chains face increasing pressure over the utilisation of scarce resources and the generation of environmentally relevant emissions, and global initiatives have been initiated to benchmark the impact of livestock supply chains on climate change (MacLeod et al., 2013) and other impacts. Life cycle assessment (LCA) has been widely used to benchmark the environmental performance of supply chains globally. However, the lack of internationally agreed methods can make comparison difficult (De Vries and De Boer, 2010.). In Australia, a series of studies investigating regional or national livestock production systems, using broadly comparable methods, have been completed by the authors and others. These studies include regional beef and lamb production (Ridoutt et al., 2012; Wiedemann et al., 2015b) and pork production (Wiedemann et al., 2016a). These studies provide a regional knowledge base for understanding the environmental impact of Australian meat production, but there is a need for more studies focussing on poultry production. Future increases in global demand for grain and meat (FAO, 2009) are expected to result in greater pressure on water and arable land. Most LCA research in Australia has focussed on greenhouse gas...
(GHG) emissions and this is an acknowledged issue of global significance. Because in Australia water is a scarce and heavily allocated resource (MDBA, 2012) and arable land represents only a small fraction of the total land mass available (Lesslie and Mewett, 2013), and further research is needed in these areas.

The Australian chicken meat industry is vertically integrated with modern, efficient production systems that aim to maximise the environmental efficiency of their production systems. Feed conversion ratio in chicken meat production is low relative to other species, resulting in lower impacts via feed production. Because of the controlled nature of production, where most birds are housed indoors, the direct impacts are also minimised. However, few data are available to quantify the performance of the industry or the contribution of impacts from each stage in the supply chain. One study (Bengtsson and Seddon, 2013) investigated the impacts of chicken meat production from a large, vertically integrated company in Australia, and a number of chicken meat LCAs have been completed elsewhere in the world (i.e. Leinonen et al., 2012; Pelletier, 2008). These studies highlight the significance of feed production as a major contributor to GHG and nutrient related impacts from chicken meat production, though in most cases a comprehensive assessment of primary resources, viz; energy, water and land was not included. These studies have shown that GHG impacts from chicken meat arise predominantly from soil nitrous oxide and fossil fuel use in crop production and housing, and manure related emissions. Most chicken meat studies (i.e. Leinonen et al., 2012; Pelletier, 2008; Williams et al., 2006) did not include impacts from meat processing, even where results are reported on a carcass weight (CW) basis, and consequently, energy and water use from this stage may have been underestimated. Because of the low input nature of Australian grain production and predominantly dry soil conditions, soil nitrous oxide and fuel use in Australian crop production may be much lower than other regions of the world, corresponding to lower feed related GHG emissions. Conversely, electricity related emissions are high in Australia because of the reliance on coal fired electricity generation, which will therefore result in higher impact from energy intensive stages in the supply chain, such as housing and meat processing in Australia. This study aimed to determine GHG, fresh water consumption, fossil energy demand and land occupation to provide a benchmark for Australian conventional and free range production, and determine impact hotspots in the supply chain, by applying methods and inventories representative of Australian production and processing. The study included two major, contrasting production regions and collected data from multiple companies, to provide results that are broadly representative of Australian chicken meat production.

2. Materials and methods

The study utilised primary and secondary data sources and methods reflecting Australian production systems where available. Specific data collection and modelling approaches are outlined in the following sections.

2.1. Impacts assessed

The study assessed GHG emissions using the IPCC AR4 global warming potentials of 25 for CH$_4$ and 298 for N$_2$O, as applied in the Australian National Inventory Report (NIR) (Commonwealth of Australia, 2015b). GHG emissions associated with land use (LU) and direct land use change (dLUC) were included and reported separately, following guidance from the Livestock Environmental Assessment and Performance partnership (LEAP, 2014). Fossil fuel energy demand was assessed by aggregating all fossil fuel energy inputs throughout the system and reporting these per mega joule (MJ) of energy, using lower heating values (LHV). Fresh water consumption (L) was assessed using methods consistent with ISO (2014), as described in the following sections. One exception was the assessment of fresh water consumption associated with land use change (LUC) which was not assessed because of the lack of suitable inventory data for the background grain processes. The impact on water use was assessed using the stress-weighted water indicator, based on Pfister et al. (2009). The value was expressed as a water equivalent (H$_2$O-e; Ridoutt and Pfister, 2010), by dividing the stress weighted water value by the global average water stress volume. Land occupation was assessed by aggregating impacts throughout the supply chain, and both total land occupation and arable land occupation are reported in square meter years (m$^2$ yr$^{-1}$). All modelling was carried out using SimaPro™ 8.0 (Pre-Consultants, 2014) and the study applied an attributional modeling approach.

Production from two Australian states (Queensland — QLD and South Australia — SA) and two production systems: conventional housing (indoor housing with tunnel ventilation), and free range (FR) production were investigated. Queensland is located in the mid-north eastern part of Australia, while SA is located in the southern-central part of the continent. Each production region mainly utilised feed produced in the region, though the QLD supply chain utilised slightly more imported feed ingredients. The primary production supply chain included breeding (rearing of parent birds, fertile egg production and hatchery processes), grow-out and meat processing, with all associated inputs. Grandparent and great grandparent breeding systems were not included since they were found to contribute <1% of impacts, in a preliminary scoping analysis conducted as part of the study (unpublished data). Data were collected as part of the study to cover a 12 month production period (2009–2010), from three major vertically integrated poultry producers across 38 facilities. The FR supply chain consisted of one company supply chain in each state, each with multiple FR farms. These were combined to ensure company data were confidential, and to provide a larger and more representative FR dataset. However, a limitation to this was that we could not compare conventional with FR production within each state supply chain. Data collection processes are described in the following sections. The end-point of the supply chain was the cold storage unit where chicken meat is stored prior to wholesale distribution. Results are presented using two functional units (FU): 1 kg of chilled chicken (whole bird) ready for packaging and distribution to retail, and 1 kg of boneless, skinless chicken portions, ready for packaging and distribution to retail. The system boundary of the study is shown in Fig. 1.

2.2. Life cycle inventory

The inventory was separated and reported separately for each stage of the supply chain. Data collection methods and calculation methods are described in the following sections.

2.2.1. Feed use and milling

Feed use for breeding birds and meat chickens was reported by each company, in each state. Birds are phase fed, and diets may change during the year in response to changes in the availability of commodities. Each company operated their own feed mill, and commodity inputs, energy and water use, and transport distances were reported by each feed mill over a 12 month period (Table 1). The aggregated rations are shown in Table 2.

2.2.2. Feed production

Major feed grains were modelled for Australian grain processes by the authors, or using processes available from the AustLCI
where available. All processes used emission factors from the Australian NIR 2013 (Commonwealth of Australia, 2015b). For major grains in each region, the proportion of grain produced in different systems (i.e. dry land and irrigated) was determined using the proportion of crop land irrigated and average irrigation rates in each state over three years reported by the ABS (ABS, 2009, 2010, 2011). Losses associated with the supply of irrigation water were 27.1%, based on the national water accounts (ABS, 2012). Grain processes were aggregated to provide an average market for the major grains in each state. LU and dLUC emissions were not included in the Australian grain inventory datasets available, and were therefore assessed separately. Annualised emissions associated with conversion of forest land to crop land were 4,755,000 t CO2-e in the period 1990–2010 (Commonwealth of Australia, 2015a). The analysis of LU emissions from crop land were −4,800,000 t CO2-e (negative emissions indicate carbon sequestration), annualised

Table 1
Average feed milling inputs per tonne of ration produced for Queensland and South Australian supply chains.

<table>
<thead>
<tr>
<th>Inputs</th>
<th>Queensland</th>
<th>South Australia</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity (kWh/tonne(^a))</td>
<td>25.4</td>
<td>18.8</td>
</tr>
<tr>
<td>LPG (MJ/tonne(^b))</td>
<td>56.4</td>
<td>0.2</td>
</tr>
<tr>
<td>Natural gas (MJ/tonne)</td>
<td>0.0</td>
<td>69.0</td>
</tr>
<tr>
<td>Fresh water consumption (L/tonne)</td>
<td>72.0</td>
<td>89.4</td>
</tr>
<tr>
<td>Transport (t.km/tonne)</td>
<td>286.1</td>
<td>105.4</td>
</tr>
</tbody>
</table>

\(^a\) Kilo watt hours, reported per tonne, “as-fed” (inclusive of moisture).
\(^b\) Mega joule.
\(^c\) Tonne kilometres.

database (Life Cycle Strategies, 2015) where available. All processes used emission factors from the Australian NIR 2013

Fig. 1. Chicken meat production system boundary. FU: functional unit.
over the same period. Carbon sequestration in Australian crop land is mostly in response to carbon sequestration resulting from adoption of improved cropping practices over the past 20 years. When divided by the average total land area sown to crops annually in Australia over the period 1990–2010, annualised emissions from LU and dLUC were 229 and 227 kg CO2-e/ha. Differences in LU emissions or sequestration may exist between cropping regions in Australia based on specific management. However, as suitable disaggregated datasets were not available to assess impacts associated with individual crops, or cropping regions by state, in the present study we accounted LU and dLUC emissions from Australian crop land at the national scale. Where data were unavailable for some small dietary inputs, such as vitamins, substitutions were made with other feed inputs using product cost to guide the substitution. Imported soybean meal was modelled using data from the EcoInvent database (Swiss Centre for Life Cycle Inventories, 2014), based on the relative imports of soybean meal from the major sources of Australian imports; South America (80%) and the United States of America (20%) (OEC, 2015). Irrigation water associated with imported soybeans after Aldaya et al. (2010), and irrigation water use in South America was assumed to be 877 m3/tonne irrigated soybeans (Aldaya et al., 2010), with irrigated soy representing 3% of the total crop.

### 2.2.3. Breeding and hatching

Inputs associated with breeding and hatching were collected from five company’s breeder farms and hatcheries across the two states. Water data were collected from farm records of the total volume of water pumped. Water was predominantly used for drinking and cleaning. Drinking water, after ingestion, was respired, excreted with manure or integrated into the bird. Each of these flows ultimately resulted in water consumption or removal from the original water catchment in the product. Thus, all drinking water was treated as fresh water consumption. Likewise, cleaning water was considered a consumptive use as small volumes were used and sheds were left to dry out after cleaning, resulting in evaporation of the water used. Where water was supplied from a system that incorporated open water storages, evaporative losses were assessed and included in the total volume of fresh water consumption used. Major inputs are shown in Table 3. In addition to inputs, the mass of live weight in spent hens for meat processing was 32.2 and 22.0 kg for QLD and SA respectively.

### 2.2.4. Breeding farm manure management

Manure GHG emissions (CH4, N2O) and indirect emission precursors (NH3) were estimated by predicting nitrogen (N) and volatile solids (VS) excretion using mass balance principles, and applying emission factors from the Commonwealth of Australia (2015b). Excreted N was determined from the difference between N inputs (in feed) and N outputs (in bird mass, mortalities and eggs). Excreted VS was determined by subtracting manure ash excreted from total solids (TS), which represented the residual of non-digested feed (Dong et al., 2006). The sensitivity of the model to manure emission factors was tested by performing a comparison with the IPCC factors (Dong et al., 2006). Manure was typically removed from the site and sold for a small amount of money, and was treated as a residual, with no allocation process applied (LEAP, 2015). Indirect nitrous oxide was modelled from ammonia volatilisation. All animal houses were constructed with impervious floors and therefore nitrate leaching and runoff were assumed to be negligible. Factors are shown in Table 4.

#### 2.2.5. Meat chicken grow-out phase

Flock performance, including feed intake, growth rate, mortality rate and the total mass of birds harvested were determined from records supplied by each company in each state and represent actual performance under commercial conditions (Table 5). The grow-out phase is generally contracted out to third party growers in Australia, and these growers are responsible for animal husbandry, housing and litter management. Records of water use, energy use, cleaning and litter management were collected from 22 farms across the two states, covering a 12 month period. Volumes of drinking, cooling and cleaning water were collected from farm records. Drinking and cleaning water were handled in the way described for the breeding flocks, whereas cooling water was used in evaporative cooling systems and was therefore a consumptive use. In some cases, on-site water storages were used and evaporation from these storages was included in the total volume of fresh water consumption used. Material inputs are shown in Table 6.

#### 2.2.6. Grow-out phase manure management

Manure excretion and manure emissions were determined using the same mass balance approach described for the breeder facilities. Emission factors for birds housed on litter were from the Commonwealth of Australia (2015b) and recent Australian research (Wiedemann et al., 2016b), with the latter given...
preference where data were available. Factors are shown in Table 4. The sensitivity of the model to manure emission factors was tested by performing a comparison with the IPCC factors (Dong et al., 2006). Manure was removed from the meat chicken houses at the end of each production batch, and sold to local farmers as a fertiliser. Manure sales represented a very small fraction of the total value of production. As a result, they were treated as a residual flow with no allocation of impacts from the production system to the manure product. Impacts from manure following removal from the chicken house were assumed to be attributed to the system using the manure as a fertiliser.

2.2.7. Meat processing
Meat processing data were averaged from data collected at three processing plants in QLD and two processing plants in SA. All meat processing plants used water from reticulated supplies, with some facilities supplementing this with on-site groundwater extraction. All but one processing plant released effluent water into the city sewerage treatment system, where it was treated and returned to the same water catchment. One QLD processing plant irrigated water to pasture on-site, which was considered a consumptive use attributable to the meat processing system. The processing plants reported resource use relative to carcass mass, but all plants produced a combination of chicken products, including (carcase weight) CW and both bone-in and boneless portions. Major inputs associated with processing are reported in Table 7.

Table 4
Manure greenhouse gas emission factors for meat chicken houses.

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Factor applied</th>
<th>Comparison factor — IPCC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manure methane, Methane Conversion Factor (MCF)</td>
<td>0.007&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.015&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Manure nitrous oxide, kg N₂O-N/kg N excreted</td>
<td>0.0035&lt;sup&gt;c&lt;/sup&gt;</td>
<td>0.001&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Manure nitrous oxide (FR area), kg N₂O-N/kg N excreted</td>
<td>0.02&lt;sup&gt;f&lt;/sup&gt;</td>
<td>0.02&lt;sup&gt;f&lt;/sup&gt;</td>
</tr>
<tr>
<td>Manure ammonia, kg NH₃-N/kg N excreted</td>
<td>0.11&lt;sup&gt;a&lt;/sup&gt;</td>
<td>0.4&lt;sup&gt;c&lt;/sup&gt;</td>
</tr>
<tr>
<td>Indirect nitrous oxide, kg N₂O-N/kg NH₃N</td>
<td>0.002&lt;sup&gt;b&lt;/sup&gt;</td>
<td>0.01&lt;sup&gt;d&lt;/sup&gt;</td>
</tr>
</tbody>
</table>

<sup>a</sup> Wiedemann et al. (2016b).
<sup>b</sup> Commonwealth of Australia (2015b).
<sup>c</sup> Dong et al. (2006).
<sup>d</sup> De Klein et al. (2006).

Table 5
Performance data for meat chickens in the grow-out phase.

<table>
<thead>
<tr>
<th>Production data</th>
<th>Queensland</th>
<th>South Australia</th>
<th>Free range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total bird production, per year</td>
<td>949,662</td>
<td>1,955,162</td>
<td>590,191</td>
</tr>
<tr>
<td>Final bird weight, kg</td>
<td>2.5</td>
<td>2.8</td>
<td>2.4</td>
</tr>
<tr>
<td>Final bird age, days</td>
<td>42.8</td>
<td>42.2</td>
<td>41.6</td>
</tr>
<tr>
<td>Flocks, per year</td>
<td>5.5</td>
<td>5.9</td>
<td>6.5</td>
</tr>
<tr>
<td>Feed conversion ratio, kg feed/kg live weight</td>
<td>1.89</td>
<td>1.85</td>
<td>1.89</td>
</tr>
</tbody>
</table>

Table 6
Grow-out phase inputs reported per 1000 kg of live weight produced.

<table>
<thead>
<tr>
<th>Materials</th>
<th>Queensland</th>
<th>South Australia</th>
<th>Free range</th>
</tr>
</thead>
<tbody>
<tr>
<td>Feed ration, kg as-fed</td>
<td>1886.0</td>
<td>1853.0</td>
<td>1886.0</td>
</tr>
<tr>
<td>Day-old chicks</td>
<td>423.4</td>
<td>402.5</td>
<td>448.0</td>
</tr>
<tr>
<td>Electricity, kWh</td>
<td>99.8</td>
<td>96.2</td>
<td>82.9</td>
</tr>
<tr>
<td>LPG, L</td>
<td>13.2</td>
<td>26.5</td>
<td>19.8</td>
</tr>
<tr>
<td>Natural gas, m³</td>
<td>25.7</td>
<td>n.a</td>
<td>n.a</td>
</tr>
<tr>
<td>Diesel, L</td>
<td>0.4</td>
<td>1.6</td>
<td>0.9</td>
</tr>
<tr>
<td>Petrol, L</td>
<td>0.6</td>
<td>0.5</td>
<td>1.2</td>
</tr>
<tr>
<td>Staff transport, km</td>
<td>9.6</td>
<td>3.7</td>
<td>3.7</td>
</tr>
<tr>
<td>Fresh water consumption – animal houses, L</td>
<td>3206.0</td>
<td>5890.0</td>
<td>5014.0</td>
</tr>
<tr>
<td>Fresh water consumption – water supply system losses, L</td>
<td>83.2</td>
<td>n.a</td>
<td>n.a</td>
</tr>
<tr>
<td>Bedding – shavings/straw, kg</td>
<td>127.0</td>
<td>162.0</td>
<td>240.5</td>
</tr>
<tr>
<td>Pesticides, L</td>
<td>0.1</td>
<td>0.1</td>
<td>0.1</td>
</tr>
<tr>
<td>Disinfectant, L</td>
<td>0.8</td>
<td>0.5</td>
<td>0.7</td>
</tr>
</tbody>
</table>

2.3. Handling Co-production
Total product mass from the system was inclusive of meat chickens, small amounts of meat from end-of-life breeding hens, and edible offal. Co-products included manure, pet food and processing by-products for rendering. Manure was a very low value output from the system and was treated as a residual. Emissions associated with transport, storage and land application of manure were attributed to the systems utilising the manure, which included grain production systems producing grain for the chicken meat system. Some 19% of litter is used for grain production in Australia (Dorahy and Dorahy, 2008), and total N supply from poultry litter contributed 1% of total crop N requirements.

Meat processing by-products (i.e. renderable products, pet
foods) were handled using economic allocation. The economic value of CW represented 98.5—99.2% of the output from meat processing, with the renderable and pet food products contributing the remaining revenue. Impacts reported per kilogram of chicken portions were determined using yield factors and mass flows described in Wiedemann and Yan (2014).

3. Results

Fossil fuel energy demand was 18.1 and 21.4 MJ/kg CW for the SA and QLD conventional systems respectively, and 18.3 MJ/kg CW from FR production. Feed production contributed 53—59% of fossil fuel energy demand, with most energy in the feed production system being related to field operations and energy associated with fertiliser manufacture. Meat processing (13—16%) and housing (18—21%) were also substantial contributors to fossil fuel energy demand.

Fresh water consumption ranged from 38 to 111 L/kg CW for the SA and QLD conventional production systems, and 70 L/kg CW from FR production. Water associated with the production of feed contributed 69—86% of fresh water consumption, predominantly from irrigation of grain. The grow-out phase used 5—21% of total water, mainly for drinking and cooling, while meat processing and breeding contributed smaller amounts.

Stress weighted water use was 26 and 36 L H2O-e/kg CW in the QLD and SA conventional supply chains respectively, and 21 L H2O-e/kg CW in the FR supply chain. Stress weighted water use was primarily associated with irrigation water use, though the contribution from the grow-out and processing stages of the supply chain tended to be higher than observed for fresh water consumption, because these operations tended to occur in more water stressed catchments. Notably, stress weighted water use was higher in the SA supply chain, despite fresh water consumption being lower, because water stress was higher in parts of this supply chain.

Arable land occupation ranged from 14.0 to 22.5 m²/kg CW for the QLD and SA conventional production systems, and 18.2 m²/kg CW for FR production. Total land occupation was 1—2% higher than arable land occupation, when the small amount of land used for bird housing, processing and other background processes were included.

Greenhouse gas emissions (excl. LU and dLUC) were 2.2 and 1.8 kg CO₂-e/kg CW for QLD and SA conventional production respectively. Emissions from FR production were 1.8 kg CO₂-e/kg (Fig. 2). Emissions from LU and dLUC were 1.2 kg CO₂-e for QLD and 1.0 kg CO₂-e/kg CW for the SA conventional systems. Impacts from FR production were 0.4 kg CO₂-e/kg CW. LU and dLUC emissions arose from soymeal production, which was included at a higher rate in the conventional diets compared to the FR. Feed production represented the largest contributing stage of the supply chain for GHG emissions, with impacts ranging from 55 to 60%, and 64—75%, if LU and dLUC were included. Emissions from the grow-out phase were 12—16%, with the majority of this being associated with energy use for housing and a smaller proportion contributed from manure emissions. Meat processing contributed 12—18%, with the main differences relating to energy use and emissions arising from waste water treatment, which were lower in the SA supply chain.

Impacts per kilogram of chicken portions are shown in Table 8. Results were 39% higher than for chilled, whole birds because of the loss of mass associated with the boneless product and the additional impacts associated with further processing.

4. Discussion

This study provides a new benchmark assessment of Australian chicken meat production through to production of a wholesale product, with novel results related to water, water stress and arable land occupation. The GHG assessment includes revised methods supported by recent research in manure management and crop fertiliser emissions, and included assessment of LU and dLUC emissions. We did not include nutrient related impacts such as eutrophication, and did not investigate land degradation impacts, and these areas require further research to produce a broader assessment of environmental impacts from the supply chain.

4.1. Sensitivity to model assumptions

Across all impact categories, feed use was the greatest source of
 impacts, and the model was therefore sensitive to a number of feed related assumptions. Impacts were greatest from the three largest commodities, wheat, sorghum (QLD) and soybean meal. The relatively low emission profile of Australian cereal grains was the major factor explaining low emissions from chicken meat in the present study. We compared grain inventory processes applied in the present study (derived from the AustLCI–Life Cycle Strategies, 2015) with other Australian studies (Brock et al., 2012) and found few differences in GHG emissions, provided emission factors were harmonised. This suggests a reasonable level of agreement between grain LCI data for major Australian processes. We found that the relative proportion of different cereals (wheat, barley and sorghum) also had little effect on the impacts generated by the diet. This was provided the overall proportion of cereals was the same, because the impacts associated with the different cereal grain productions systems was similar. However, differences in the inclusion rate of soybean meal resulted in substantial differences in dLUC emissions, fossil fuel energy demand, fresh water consumption and stress weighted water use. Where diets were formulated to utilise Australian field pea as the major protein grain, impacts were found to be 15% lower for GHG and energy, 90% lower for fresh water consumption, 90% lower for stress weighted water use and 94% lower for LU and dLUC emissions. This dietary change was more extreme than those modelled by Leinonen et al. (2013) and the change in LU and dLUC emissions was consequently much greater. In practice, smaller reductions in soymeal inclusion are more likely. The diets modelled in the present study were representative of the vast majority of chicken meat production in each state over the 12 month period assessed. However, inter-annual variation in commodity inclusion rates may occur and these results may only be taken as representative of diets with similar inclusion rates of soymeal.

Fresh water consumption and stress weighted water use were also sensitive to assumptions regarding irrigation rate and irrigation region. From year to year, water supply fluctuates substantially and consequently irrigation rates and the total area irrigated will vary. In the present study, a three year average irrigation rate was applied and the total irrigation volume was divided by total crop yield in each state, to determine the weighted average volume of irrigation water per tonne of grain produced. The dataset did not allow irrigation water flows to be attributed to specific cereals or end markets (human consumption vs animal consumption), and this remains a knowledge gap to be addressed by future research. To explore the sensitivity of the model to inter-annual variation in irrigation rate, we compared irrigation rates from a low water availability year (2010) and a high water availability year (2008). For the state with the largest volume of irrigation use (QLD), water was found to vary between 83 and 157 L/kg CW between the two years. Inter-year variation in stress weighted water use is expected to be very high in response to the different rates of irrigation used and the variable rates of extraction (and therefore water stress) from year to year. In the present study, static, coarse resolution water stress values were used and as a consequence, stress weighted water use results should be viewed with a degree of caution. These results could be improved through development and application of annual WSI values for Australian catchments.

Greenhouse gas emission results per kilogram of chicken were found to be ~4% higher when IPCC (Dong et al., 2006) emission factors were applied, suggesting the model was not sensitive to the application of emissions from Australian research. To test the sensitivity of the results to electricity use, these inputs were varied by 15% (difference between the highest and lowest company average) which resulted in <1% impact on total fossil energy. Differences in electricity use during meat processing resulted in more significant changes in total fossil energy, with more efficient plants having 5% lower total energy per kilogram of chicken than less efficient plants.

### 4.2. Main impact sources and mitigation

Australian meat chicken production is dominated by large, vertically integrated producers that manage the genetics, nutrition and processing of the birds, resulting in benefits from economies of scale and high degrees of efficiency at critical points in the supply chain. The Australian industry is focused primarily on domestic production rather than export, and has grown at an annual rate of ~2.5% each year for the past 10 years (ABS, 2014), in response to increased consumer demand for product. However, as the chicken meat industry continues to expand, the impact on finite resources and impacts from production will also increase. As a result, production efficiency throughout the supply chain remains a priority, particularly in the areas of greatest impact.

While the industry has no direct control over grain production, it represents the major impact area for arable land and water resources, and the scarcity of these resources will be relayed to chicken meat through grain supply. While some opportunities may exist to change the type of grains used by the industry to reduce impacts, the long term efficiency will be more heavily influenced by changes in feed conversion ratio (FCR). According to McKay et al. (2000), the annual rate of improvement in FCR is 0.02, resulting in a 0.6 kg reduction in feed requirements over the previous 30 years. In the present study, a 0.1 improvement in FCR resulted in a 3−4.5% decrease in GHG (inc. LU and dLUC) depending on the diet. Improved FCR has a positive effect on both upstream impacts from feed production, and downstream impacts from manure emissions, as manure production decreases with improved FCR. Wiedemann et al. (2016b) demonstrated that reduced dietary crude protein, achieved by improving the balance of amino acids in the diet to ensure the nutritional requirements of the bird was optimised, could reduce both FCR and manure emissions. This could result in lower impacts from two major impact hotspots in the supply chain. For those supply chains with water intensive diets (QLD), improved feed conversion ratio also reduced fresh water consumption by as much as 5% for each 0.1 change in FCR. Similarly, improved FCR reduced arable land requirements, particularly in SA where yields are lower and land requirements are consequently higher. Such improvements are vital for the ongoing sustainability of the industry, to minimise

### Table 8

<table>
<thead>
<tr>
<th></th>
<th>Fossil fuel energy demand, MJ</th>
<th>Fresh water consumption, L</th>
<th>Stress weighted water, Arable land, m²</th>
<th>Greenhouse gases, excl. LU and dLUC, kg CO₂-e</th>
<th>Greenhouse gases, LU an dLUC, kg CO₂-e</th>
</tr>
</thead>
<tbody>
<tr>
<td>Queensland</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>conventional</td>
<td>29.8</td>
<td>154.7</td>
<td>36.4</td>
<td>19.5</td>
<td>3.1</td>
</tr>
<tr>
<td>South Australia</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>conventional</td>
<td>23.1</td>
<td>52.7</td>
<td>50.7</td>
<td>31.3</td>
<td>2.5</td>
</tr>
<tr>
<td>Free range</td>
<td>25.5</td>
<td>96.8</td>
<td>29.0</td>
<td>25.3</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td>0.56</td>
</tr>
</tbody>
</table>

*Land use.

Direct land use change.
the pressure on finite water and arable land resources.

After feed use, the grow-out phase and meat processing contributed the largest proportion of GHG, energy and fresh water consumption in the supply chain. The substantial contribution from the grow-out phase to GHG and energy demand has prompted further research into energy efficiency from the grow-out stage (McGahan et al., 2012), and the companies participating in the study have implemented energy and water saving activities in the meat processing plants to reduce impacts over time. Modest opportunities exist to mitigate manure GHG, though these strategies are not expected to have a substantial impact on the GHG emissions from full supply chain. Further research into energy production from manure/litter may provide greater opportunities to reduce emissions and lower energy demand from production, and further analysis is warranted in this area.

4.3. Regional differences

Between the different regions assessed, production in SA tended to be more energy efficient, with lower GHG emissions and lower fresh water consumption. This result was due, in part, to lower supply chain fossil fuel energy demand in SA, and mainly because the diets used in SA had ~20% lower GHG and energy impacts and 63% lower fresh water consumption than the QLD diets. In contrast to this, arable land occupation was higher, as a result of the lower intensity grain production systems and lower yields in this region compared with the major grain production regions in QLD and northern New South Wales (NSW). Stress weighted water use was also higher, because the SA supply chain was more reliant on highly stressed catchments. As a result, the impact of water use in SA was greater than QLD despite the lower volume of water used. Considering the importance of location in assessing impacts from feed production, an analysis of production in the major state of NSW was done for comparison. Impacts from this supply chain were comparable to the QLD supply chain for all impacts except water stress, which was 98 L H2O-e/kg CW in response to highly stressed irrigation water use in southern NSW. Considering this finding, we expect that the national average fresh water consumption to be closer to the QLD value than SA, and water stress to be higher than observed in either state studied here. Similarly, national water use estimates for Australian beef were higher than regional estimates (Wiedemann et al., 2015a). These results, while quite specific to Australia, suggest that water may be a much more variable input between states, and possibly countries, than GHG. Accurate benchmarking would require consistent, accurate, spatially and temporally specific feed grain datasets to produce robust results.

4.4. Benchmarking

Comparison of LCA studies is complicated by the application of different assessment methods, system boundaries and assumptions. Per kilogram of live weight, GHG emissions excluding LU and dLUC in the present study ranged from 1.1 to 1.3 kg CO2-e, which was similar to production in the USA (Pelletier, 2008), but lower than previously reported by Bengtsson and Seddon (2013) for Australian chicken meat production. With LU and dLUC emissions included, results were 2.2 and 1.9 kg CO2-e/kg live weight (LW) for the conventional production and 1.6 kg CO2-e/kg LW for FR, which was lower than Leinonen et al. (2012) who reported values of ~3.1–3.6 kg CO2-e when values were converted to a LW basis. The main factors contributing to lower emissions in the present study relate to feed sources and to some extent, manure emissions. GHG emissions associated with rations in the present study ranged from 0.39 to 0.48 kg CO2-e/kg feed, or 0.53–0.92 kg CO2-e/kg ration with LU and dLUC emissions included. By comparison, Leinonen et al. (2012) reported impacts of 1.1 and 1.0 kg CO2-e for their standard and FR rations respectively, and impacts were 1.5 kg CO2-e/kg feed in MacLeod et al. (2013). The lower impacts associated with Australian feed production are well understood. Australian emission factors for crop fertiliser application (Commonwealth of Australia, 2015b) are 80% lower than international defaults reported by De Klein et al. (2006), because of the low soil moisture conditions experienced in Australia. While yields are also low, energy related inputs relative to grain production are low compared to northern Hemisphere grain production. As a consequence, impacts for major Australian grains such as wheat are in the order of 0.2 kg CO2-e/kg grain (i.e. Brock et al., 2012) and these values are lower still with more recent fertiliser emission factors applied (Commonwealth of Australia, 2015b). Field emissions associated with manure application to crops, which were substantial in MacLeod et al. (2013), were found to be minor in Australia, where manure is a small source of crop fertiliser N. The present study also provided updated emissions from manure which were substantially lower than previously reported by Bengtsson and Seddon (2013) for Australian chicken meat, as this study used superseded inventory emission factors.

Fossil fuel energy demand in the present study was 11.5–13.1 MJ/kg LW for the conventional production systems, which was lower than the 18 MJ reported by Leinonen et al. (2012), but similar to the 14.9 MJ/kg LW reported by Pelletier (2008) for broiler production in the USA. No studies were found that reported water use using a comprehensive assessment of fresh water consumption. Never-the-less, results were of a similar magnitude to the fresh water depletion reported by Bengtsson and Seddon (2013), and much higher than the 3–5 L/kg LW of direct water use (inventory value) reported by Leinonen et al. (2012), though this study does not appear to have considered fresh water consumption associated with feed production. Fresh water consumption and stress weighted water may be higher in Australia compared to northern Hemisphere countries, because of the requirements for cooling in chicken meat houses, and irrigation of crops. Land occupation was also substantially higher than reported by Leinonen et al. (2012), in response to the much lower yields from Australian crop production.

In the present study, we found FR production to perform similarly to conventional production with respect to GHG emissions and fossil energy demand, which is contrary to results of Leinonen et al. (2012) and Williams et al. (2006), who found FR to have higher emissions. The similar results between conventional and FR production in the present study were not surprising, considering the similar level of feed conversion efficiency and housing impacts in the two systems. In contrast to the GHG and energy results, water and arable land results for the FR production system were intermediate between the QLD and SA results, primarily in response to the diet composition of the particular FR systems modelled. As the FR farms were located in both states, the proportion of QLD and SA ration components was approximately equal, leading to intermediate impacts for water and arable land; factors that varied strongly between the two states.

This study is the first LCA of Australian chicken meat applying comparable methods to comparison studies for other major Australian livestock products; beef, lamb and pork. When compared per kilogram of boneless product, the study revealed much lower GHG emissions for chicken meat than beef or lamb (Wiedemann et al., 2015b) and around half the emissions of Australian pork (Wiedemann et al., 2016a). However, energy demand was similar to the 27 MJ per kg for grass-fed beef and 24 MJ per kg of lamb, when both were reported at the processor gate. Arable land occupation was also much higher than the 3.2 m² reported for grass-fed beef, and 2.5 m² reported for lamb. Unsurprisingly, total land occupation was much higher for beef and lamb
compared to chicken meat, because of the large areas of non-arable rangelands used for grazing, though the total area of rangelands in Australia is less constrained than the area of arable land. Fresh water consumption was considerably lower for chicken meat compared to beef, though stress weighted water use showed less of a difference between the two. However, because of the regional specificity of stress weighted water assessment, a national average would be required to understand impacts of one industry relative to another with respect to water use.

5. Conclusions

Consistent benchmarking of impacts from meat production systems is hampered by differences in system boundaries, changes in research methods and variation in datasets. This study provides new findings for Australian chicken meat using improved methods, inventories and assumptions to benchmark key resources and environmental impacts. This study presents one of the first comprehensive assessments of water consumption and water impacts associated with chicken meat production. Fresh water consumption (38–111 L/kg CW) was found to be substantially higher than stress weighted water use (21–36 L H2O-e/kg CW), highlighting the importance of water impact assessment. Water consumption was much more variable between the different production states than GHG, and water consumption also varied substantially from year to year, as a result of changes in irrigation water availability. As a consequence, spatially and temporally specific datasets are required for accurate benchmarking of these resources in Australia. Further studies incorporating fresh water and stress weighted water use in other regions of the world are required to expand the knowledge base regarding water use in meat chicken production. The study found that total GHG impacts were low, ranging from 2.2 to 3.4 kg CO2-e/kg CW, including LU and dLUC emissions, though inclusion of meat processing increased these emissions by up to 8% compared to the more common farm-gate analyses. Considering feed production and use was the major impact source for all categories studied, improved FCR will result in lower impacts, provided changes don’t occur to the diets, or feed supply chains. Impacts were similar between conventional and free range production, with the observed differences primarily related to difference in diets rather than differences in housing. This study did not include nutrient related impacts such as eutrophication or acidification, and did not investigate impacts on land degradation from cropping. Provided suitable methods and datasets are available, inclusion of these impacts is required to broaden the knowledge base regarding impacts from Australian chicken meat. Considering the variability found between states, expansion of this study to include more regions would provide an improved understanding of total impacts and impact hotspots in Australia. Considering the ongoing improvement in feed conversion ratio and production efficiency, and the development of new methods and knowledge in areas covered by this research, revision is recommended at regular intervals to highlight change over time.

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This study was funded by the Rural Industries Research and Development Corporation (RIRDC) Meat Chicken Program, and research was performed by FSA Consulting. The participating companies, growers and facility managers are thanked for supplying data, and Mr Glenn Poad and Dr Mingjia Yan are thanked for assistance with data collection and collation.

References

cycle energy use and greenhouse gas, ozone depleting, acidifying and eutro-phying emissions. Agric. Syst. 98, 67–73.
8.3 ADDENDUM

The system boundaries applied in this study were consistent with the summary tables presented in the discussion (Table 4 and Table 5) and no modifications were required. Human edible protein required (HEPR) was not reported in this study and was calculated to produce Figure 3, section 10.5.7. The HEPR was determined by calculating the percentage of human edible protein in the diet of the meat chickens and breeders, relative to one kilogram of boneless chicken meat produced. The weighted average percentage of diet protein was 17.8%, and total kilograms of grain per kilogram of boneless chicken was 4.6 kg, resulting in human edible protein of 0.82 kg per kilogram of boneless chicken. The protein percentage of boneless chicken was assumed to be 19% and HEPR was 4.30.

Average GHG emissions reported in Figure 3 in section 10.5.7 were determined by averaging the QLD and SA conventional results, inclusive of LU and dLUC emissions.
9 ENVIRONMENTAL IMPACTS AND RESOURCE USE FROM AUSTRALIAN PORK PRODUCTION DETERMINED USING LIFE CYCLE ASSESSMENT:

9.1 PAPER 1. GREENHOUSE GAS EMISSIONS

9.1.1 ATTRIBUTION STATEMENT

The paper *Environmental impacts and resource use from Australian pork production assessed using life cycle assessment: 1. Greenhouse gas emissions* was led by S.G Wiedemann and co-authored by E.J McGahan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
- Development and application of the greenhouse gas model
- Development of the water use model
- Development and application of grain processes used in the modelling of feed inputs
- Data acquisition and data analysis, including impact assessment
- Allocation and uncertainty analysis
- Preparation of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model and herd modelling

Murphy contributed to the study and manuscript via:

- Assistance with data analysis and assistance with preparation of figures, tables and manuscript
9.1.2 INTRODUCTION

The pork GHG emissions study (paper 1) and resource use paper (paper 2) were the final output from a series of pork industry LCA, benchmarking and GHG emission projects, initiated in 2008. This introduction covers both papers. The results were first released in grey literature reports (Wiedemann et al., 2010b; Wiedemann et al., 2012a - see electronic supplementary materials) and these analyses were later updated and expanded for the published papers presented here. Two companion studies completed by the author and colleagues (McGahan et al., 2016; Phillips et al., 2016) were the first to measure manure production and GHG emissions from Australian piggeries, providing a robust basis for the modelling analysis used in the LCA presented here. An associated study was also published outlining methods and inventory data for meat processing (Wiedemann and Yan, 2014). See supplementary materials.

This study included analysis of a range of production systems and regions and was expanded to include a national analysis of impacts for the publication. The study provides important insights into GHG emissions from different manure management systems and was used as the basis for expanding the number of ERF methods available for piggeries.

This study is one of the only LCA water use assessments for pork globally and provided a robust basis for understanding impacts and trade-offs associated with water, energy and land occupation from pork production.
Environmental impacts and resource use from Australian pork production assessed using life-cycle assessment.

1. Greenhouse gas emissions

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Abstract. Agricultural industries are under increasing pressure to measure and reduce greenhouse gas emissions from the supply chain. The Australian pork industry has established proactive goals to improve greenhouse-gas (GHG) performance across the industry, but while productivity indicators are benchmarked by industry, similar data have not previously been collected to determine supply chain GHG emissions. To assess total GHG emissions from Australian pork production, the present study conducted a life-cycle assessment of six case study supply chains and the national herd for the year 2010. The study aimed to determine total GHG emissions and hotspots, and to determine the mitigation potential from alternative manure treatment systems. Two functional units were used: 1 kg of pork liveweight (LW) at the farm gate, and 1 kg of wholesale pork (chilled, bone-in) ready for packaging and distribution. Mean GHG emissions from the case study supply chains ranged from 2.1 to 4.5 kg CO\textsubscript{2}-e/kg LW (excluding land-use (LU) and direct land use-change (dLUC) emissions). Emissions were lowest from the piggeries that housed grower-finisher pigs on deep litter and highest from pigs housed in conventional systems with uncovered anaerobic effluent ponds. Mean contribution from methane from effluent treatment was 64\% of total GHG at the conventional piggeries. Nitrous oxide arose from both grain production and manure management, comprising 7–33\% of the total emissions. The GHG emissions for the national herd were 3.6 kg CO\textsubscript{2}-e/kg LW, with the largest determining factor on total emissions being the relative proportion of pigs managed with high or low emission manure management systems. Emissions from LU and dLUC sources ranged from 0.08 to 0.7 kg CO\textsubscript{2}-e/kg LW for the case study farms, with differences associated with the inclusion rate of imported soybean meal in the ration and feed-conversion ratio. GHG intensity (excluding LU, dLUC) from the national herd was 6.36 ± 1.03 kg CO\textsubscript{2}-e/kg wholesale pork, with the emission profile dominated by methane from manure management (50\%), followed by feed production (27\%) and then meat processing (8\%). Inclusion of LU and dLUC emissions had a minor effect on the emission profile. Scenarios testing showed that biogas capture from anaerobic digestion with combined heat and power generation resulted in a 31–64\% reduction in GHG emissions. Finishing pigs on deep litter as preferred to conventional housing resulted in 38\% lower GHG emissions than conventional finishing.

Additional keywords: agricultural systems, global climate change, manure, methane, nitrous oxide.

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Introduction

In Australia and globally, there is increased interest from industries, governments and consumers regarding the environmental performance of agri-food production systems. Of these impacts, greenhouse-gas (GHG) emissions are of particular concern to governments and the general public. The Australian pork industry has established proactive goals to reduce GHG emission intensity of pork production. However, while productivity and business indicators are numerous and regularly measured, GHG emissions have not previously been benchmarked by the industry. Comprehensive assessment methods such as life-cycle assessment (LCA) include emissions throughout the production system and also emissions associated with input commodities such as feed. Such tools are effective in comparing different management systems because impacts from the whole system are taken into account. Impacts are generally reported relative to production (i.e. per kilogram of product) and, therefore, take into account the positive effect that changes in system efficiency may have on environmental indicators such as GHG emissions. LCA has been applied to determine supply chain GHG emissions at the regional or national scale for milk and dairy products (Gollnow et al. 2014), beef (Wiedemann et al. 2015a, 2016a), export lamb (Wiedemann et al. 2015b, 2016b) and chicken meat (Wiedemann et al. 2012b). The present study provides the first case study and national analysis of GHG emissions from Australian pork production using LCA. The study aimed to benchmark GHG emissions throughout the primary-production supply chain, determine emission hotspots and investigate the mitigation potential of several alternative manure-management systems.
Materials and methods

Goal and scope
The study was an attributional investigation of pork production from major production regions and different production systems in Australia, to provide information to the pork industry, research community and the general public.

The study investigated GHG emissions using the Intergovernmental Panel on Climate Change (IPCC) AR4 global warming potentials of 25 for CH4 and 298 for N2O (Solomon et al. 2007) as applied in the Australian National Inventory Report (NIR; Commonwealth of Australia 2015a). GHG emissions associated with land use (LU) and direct land-use change (dLUC) were included and reported separately, following guidance from the Livestock Environmental Assessment and Performance partnership (LEAP 2014a). The primary production supply chain, including breeding through to finishing (sometimes at multiple sites) and meat processing was included, with all associated inputs. Data were collected and impacts were assessed from a series of case-study farms (CSFs), and a national assessment was performed using national survey statistics, as described in the following sections. The end-point of the supply chain was the cold storage unit where pork is stored before wholesale distribution. Results are presented using two functional units: 1 kg of pork liveweight (LW) at the farm gate, and 1 kg of chilled, bone-in pork cuts ready for distribution to retail. The system boundary of the study is shown in Fig. 1 with the dashed line denoting the foreground system. The red arrow represents the flow of gilts (young females) and boars back into the breeding herd.

Case-study farms
Fourteen farms were surveyed in the following four major production regions of the country: Queensland (Qld), New
South Wales (NSW), Western Australia (WA) and Victoria (Vic.). Data were collected from detailed company records via site visits. A minimum of 12-month production data were collected from each farm in the Years 2010–2011 for all farms except the Vic. CSF, where data were collected in 2013. The farms were grouped into supply chains representing either an aggregation of small farms, or a series of farms (breeder, grower, finishers). The supply chains are described by state, piggery size and housing type as follows: Qld small–medium conventional (Qld SMC), Qld large conventional (Qld LC), NSW conventional housing (breeding pigs) and deep-litter housing for grower–finisher pigs (NSW C–DL), Victorian large conventional (Vic. LC), Western Australian large conventional (WA LC) and WA outdoor housing (breeder pigs) and deep-litter housing for grower–finisher pigs (WA O–DL). Each supply chain included multi-site production under the same management, with the exception of the Qld SMC supply chain, which included five independently owned, closed herd farrow-finish piggeries of 100–600 sows. Average performance data for each of the supply chains are presented in Table 1, based on farm records. Herd productivity across the CSF farms was within the range reported for the industry for pigs weaned per sow per year and feed-conversion ratio (FCR), suggesting that these were reasonably representative of industry performance (APL 2012).

National herd
To determine impacts from the Australian national herd, a separate model was developed using national herd statistics for the year 2010. Herd numbers were accessed from the survey of Australian farms (ABS 2012), which included breeder and grower pig numbers by state. An independent dataset of the total number of pigs slaughtered, and total carcass weight (ABS 2014a, 2014b), was available to determine the total output of the herd. The number of pigs produced for slaughter per sow was determined by dividing total slaughter numbers by total sow numbers. From this, litters per sow per year and pigs weaned per sow per year (Table 1) were determined using mortality rates from the CSF dataset. These productivity factors corresponded well to industry statistics (APL 2012), although FCR was slightly higher from our analysis. Breeder mortality and replacement rates in the national herd were determined from data averaged across the CSF dataset. Production was concentrated in the following four regions: NSW–Vic. (combined) (49% of production), Qld (23%), WA (11.4%) and SA (16.6%). The small Tasmanian herd (0.6% of total) was modelled using data from NSW–Vic. in the absence of specific data for this region.

The proportion of pigs across the national herd housed and managed in different housing types influenced energy use and the type of manure-management system (MMS), both being contributing factors to GHG emission intensity. These data were taken from the Australian NIR (Commonwealth of Australia 2015a), which was revised in 2014 on the basis of an industry survey of management practices. Uncertainty for the herd-production data was added to parameters that have the greatest effect on herd efficiency, on the basis of the range observed in the CSF dataset, assuming normal distributions.

Operation inputs
Operation inputs including energy, administration, veterinary and other services were accounted and reported per 100 kg LW sold (Table 2). Energy data were collected from each farm averaged over a minimum 12-month period. Purchased services (e.g. administration, veterinary services, vehicle repairs) were modelled on the basis of expenditure, using economic input–output data (Rebitzer et al. 2002). Capital infrastructure (i.e. buildings) and machinery were excluded on the basis of their minor contribution (<1% of impacts) assessed during the scoping phase. Impacts from packaging of wholesale pork were also excluded. Impacts generated off-farm via the use of purchased inputs were modelled using background data sourced from the Australian life-cycle inventory database (Life Cycle Strategies 2015) where available or the European EcoInvent (3.1) database (Weidema et al. 2015). Where measured inventory data were supplied from the CSFs, uncertainty in the estimates was assumed to be negligible. However, where estimates or calculation methods were used to determine inventory inputs or outputs, uncertainty data were included using triangular distributions, with the mean being the selected value, and the outer bounds being twice the standard deviation. Where the lower

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Unit</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C–DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O–DL</th>
<th>National herd (mean and uncertainty presented as a % of mean)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Litters/sow.year</td>
<td></td>
<td>2.1</td>
<td>2.4</td>
<td>2.3</td>
<td>2.3</td>
<td>2.3</td>
<td>2.1</td>
<td>2.3 ± 3.5</td>
</tr>
<tr>
<td>Pigs weaned/sow.year</td>
<td></td>
<td>19.1</td>
<td>22.4</td>
<td>20.0</td>
<td>24.7</td>
<td>24.7</td>
<td>16.5</td>
<td>21.4 ± 9.4</td>
</tr>
<tr>
<td>No. of sows</td>
<td></td>
<td>318</td>
<td>8229</td>
<td>4680</td>
<td>1950</td>
<td>6370</td>
<td>1956</td>
<td>231 647 ± 4.0</td>
</tr>
<tr>
<td>Sow mortality rate</td>
<td>%</td>
<td>6</td>
<td>9</td>
<td>5</td>
<td>5</td>
<td>10</td>
<td>10</td>
<td>10 ± 1.8</td>
</tr>
<tr>
<td>Sow culling rate</td>
<td>%</td>
<td>33</td>
<td>32</td>
<td>46</td>
<td>36</td>
<td>31</td>
<td>41</td>
<td>37 ± 4.0</td>
</tr>
<tr>
<td>Average sale weight of slaughter pigs</td>
<td>kg</td>
<td>104.1</td>
<td>84.2</td>
<td>99</td>
<td>107</td>
<td>96.7</td>
<td>102.3</td>
<td>97.4 ± 5.1</td>
</tr>
<tr>
<td>Average age of slaughter pigs</td>
<td>days</td>
<td>155.2</td>
<td>136.5</td>
<td>156</td>
<td>154</td>
<td>160.2</td>
<td>176.3</td>
<td>151.4 ± 5.5</td>
</tr>
<tr>
<td>FCR (whole herd)</td>
<td></td>
<td>3.2</td>
<td>2.9</td>
<td>2.7</td>
<td>2.4</td>
<td>2.7</td>
<td>3.2</td>
<td>3.1 ± 6.7</td>
</tr>
</tbody>
</table>

aData were averaged independently across the five piggeries.

bWeighted average of all pigs sold.
bound produced a negative number, this was replaced by zero (see Table 2). Purchased inputs for the national herd were determined from an inventory of 33 piggeries (FSA Consulting, unpubl. data) and the CSF inventory dataset. Uncertainty was determined from the mean, standard deviation and standard error of these datasets (see Table 2).

**Transport**

Transport data were collected for all transfers of materials within the supply chain. Major transport stages included grain transport (farm-storage, feed-mill, pig farm) and transport of pigs between farms (at multiple site facilities) and to the meat-processing plant. Transport data were calculated as tonne kilometres and were classified according to truck type, using AustLCI transport-unit processes. Staff transport to and from work was calculated from staff records and typical travel distances, and was included in the model (see Table 3).

**Feed use, feed production and milling**

Feed use at the CSFs was determined from records of feed deliveries over a 12-month period. Several diets were fed at each farm to different classes of pigs, and diets were aggregated for the breeder, weaner and grower-finisher herds to produce three simplified rations (Table 4). Feed intake and feed wastage for the national herd was determined using the PIGBAL model (Skerman et al. 2015), using herd performance data shown in Table 1. Four standard diets were modelled for the main production regions in the national herd, after Skerman et al. (2014). Diet A was considered representative of the NSW–Vic. region and Diet B was used for the Qld region. Diet D was considered representative of the WA region; however, the mung bean diet component was replaced by lupins, which was more representative of data collected by the authors from major WA piggeries (Table 4; S. G. Wiedemann, unpubl. data). Diet D was also considered representative of the SA region; however, the mung bean component was replaced by field peas, which was more representative of data collected by the authors from major piggeries in this region (S. G. Wiedemann, unpubl. data). In addition, the proportions of barley and wheat were modified to include a larger fraction of barley in the SA diet than in the WA diet. Aggregated, simplified rations for the breeder, weaner and grower-finisher units for the CSFs and the national herd are shown in Tables 4 and 5.

---

### Table 2. Aggregated general services and energy inputs for case-study farms (CSFs) and national herd per 100 kg of liveweight (LW) sold

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C–DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O–DL</th>
<th>National herd (mean ± uncertainty)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Materials</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Purchased feed (kg/100 kg LW)</td>
<td>324.6</td>
<td>289.9</td>
<td>264.9</td>
<td>240.8</td>
<td>263.0</td>
<td>321.9</td>
<td>313.5 ± 26.7</td>
</tr>
<tr>
<td>StrawB</td>
<td>n.a.</td>
<td>n.a.</td>
<td>23.1</td>
<td>n.a.</td>
<td>n.a.</td>
<td>69.9</td>
<td>18.9 ± 6.2</td>
</tr>
<tr>
<td><strong>Energy inputs</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Diesel (L/100 kg LW)</td>
<td>0.56</td>
<td>0.51</td>
<td>0.23</td>
<td>0.79</td>
<td>0.32</td>
<td>1.10</td>
<td>0.41 ± 0.29</td>
</tr>
<tr>
<td>Petrol (L/100 kg LW)</td>
<td>0.37</td>
<td>0.22</td>
<td>0.10</td>
<td>0.11</td>
<td>1.25</td>
<td>0.16</td>
<td>0.1 ± 0.08</td>
</tr>
<tr>
<td>LPG (L/100 kg LW)</td>
<td>0.27</td>
<td>0.28</td>
<td>1.94</td>
<td>0.17</td>
<td>0.17</td>
<td>0.35</td>
<td>0.2 ± 0.19</td>
</tr>
<tr>
<td>Electricity (kWh/100 kg LW)</td>
<td>15.45</td>
<td>22.21</td>
<td>16.83</td>
<td>22.03</td>
<td>20.97</td>
<td>2.95</td>
<td>16.0 (6.3–26.5)C</td>
</tr>
<tr>
<td><strong>Administrative and financial services</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Accounting, auditing and book keeping (AUS/100 kg LW)</td>
<td>1.5</td>
<td>1.6</td>
<td>1.4</td>
<td>1.7</td>
<td>1.7</td>
<td>2</td>
<td>1.7 ± 0.09</td>
</tr>
<tr>
<td>Automotive repairs (AUS/100 kg LW)</td>
<td>2.7</td>
<td>2.7</td>
<td>2.5</td>
<td>3.3</td>
<td>3</td>
<td>8.3</td>
<td>3.3 ± 1.04</td>
</tr>
<tr>
<td>Veterinary products and services (AUS/100 kg LW)</td>
<td>6.4</td>
<td>7.6</td>
<td>7</td>
<td>8.3</td>
<td>8.5</td>
<td>12.4</td>
<td>8.3 ± 0.89</td>
</tr>
</tbody>
</table>

AThe uncertainty is reported as the 95% confidence interval, based on the mean, standard deviation and standard error. Values were assumed to follow a normal distribution.

BDeep-litter pigs only.

CRange in electricity values produced a positively skewed distribution, meaning that the s.d. gives no information on the asymmetry. Hence, the first and third quartiles were used as the upper and lower bounds of the range. Values were assumed to follow a triangular distribution.

### Table 3. One-way transport distance used for livestock and purchased inputs for case-study farms (CSFs) and national herd

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C–DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O–DL</th>
<th>National herd (mean ± uncertainty)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Distance staff travel to farm (km)</td>
<td>18</td>
<td>47</td>
<td>11</td>
<td>13</td>
<td>37</td>
<td>50</td>
<td>22 ± 9</td>
</tr>
<tr>
<td>Distance straw from supplier to farmA (km)</td>
<td>n.a.</td>
<td>n.a.</td>
<td>100</td>
<td>n.a.</td>
<td>n.a.</td>
<td>70</td>
<td>85 ± 29</td>
</tr>
<tr>
<td>Distance from feedmill to farm (km)</td>
<td>103</td>
<td>115</td>
<td>55</td>
<td>28</td>
<td>75</td>
<td>75</td>
<td>85 ± 49</td>
</tr>
<tr>
<td>Distance to abattoir (km)</td>
<td>72</td>
<td>147</td>
<td>120</td>
<td>750</td>
<td>226</td>
<td>400</td>
<td>200 ± 138</td>
</tr>
</tbody>
</table>

ADeep-litter and outdoor production systems only.
Table 4. Aggregated diets per tonne of ration for the case-study piggeries

For ration component, protein contents are given in parentheses. B, breeder ration; W, weaner ration; G–F, grower-finisher ration. See text for explanation of the case-study farm codes (Qld SMC, Qld LC, NSW C–DL, Vic. LC, WA LC and WA O–DL).

<table>
<thead>
<tr>
<th>Ration component (kg)</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C–DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O–DL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>BWG–F</td>
<td>BWG–F</td>
<td>BWG–F</td>
<td>BWG–F</td>
<td>BWG–F</td>
<td>BWG–F</td>
</tr>
<tr>
<td>Barley (11%)</td>
<td>223.6</td>
<td>33.9</td>
<td>41.3</td>
<td>416.8</td>
<td>0.0</td>
<td>45.7</td>
</tr>
<tr>
<td>Triticale (17.5%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Maize (8%)</td>
<td>17.3</td>
<td>26.9</td>
<td>23.4</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Sorghum (11%)</td>
<td>232.5</td>
<td>89.6</td>
<td>296.4</td>
<td>267.3</td>
<td>37.7</td>
<td>468.5</td>
</tr>
<tr>
<td>Wheat (11%)</td>
<td>366.4</td>
<td>617.5</td>
<td>419.9</td>
<td>754.4</td>
<td>232.5</td>
<td>687.6</td>
</tr>
<tr>
<td>Lupins (34%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Milkrun (17%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>10.7</td>
<td>0.0</td>
</tr>
<tr>
<td>Bloodmeal (85%)</td>
<td>4.7</td>
<td>15.3</td>
<td>4.4</td>
<td>6.2</td>
<td>27.5</td>
<td>1.0</td>
</tr>
<tr>
<td>Meat and bone meal (50%)</td>
<td>38.7</td>
<td>42.4</td>
<td>36.3</td>
<td>40.4</td>
<td>46.7</td>
<td>56.9</td>
</tr>
<tr>
<td>Canola meal (34%)</td>
<td>33.4</td>
<td>9.8</td>
<td>93.2</td>
<td>43.2</td>
<td>0.0</td>
<td>123.0</td>
</tr>
<tr>
<td>Soymeal (48%)</td>
<td>21.4</td>
<td>105.6</td>
<td>53.2</td>
<td>17.5</td>
<td>94.2</td>
<td>46.8</td>
</tr>
<tr>
<td>Sunflower meal (30%)</td>
<td>17.1</td>
<td>0.0</td>
<td>3.1</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Other protein meal</td>
<td>5.4</td>
<td>24.0</td>
<td>0.0</td>
<td>5.0</td>
<td>25.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Canola oil</td>
<td>10.4</td>
<td>13.6</td>
<td>7.4</td>
<td>5.8</td>
<td>5.0</td>
<td>3.1</td>
</tr>
<tr>
<td>Low-cost additives</td>
<td>17.6</td>
<td>3.5</td>
<td>12.7</td>
<td>16.9</td>
<td>2.0</td>
<td>8.9</td>
</tr>
<tr>
<td>High-cost additives</td>
<td>11.5</td>
<td>18.0</td>
<td>8.7</td>
<td>10.2</td>
<td>7.6</td>
<td>4.4</td>
</tr>
<tr>
<td>Total (kg)</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
</tr>
<tr>
<td>Diet digestibility (%)</td>
<td>81.9</td>
<td>86.9</td>
<td>85.0</td>
<td>83.1</td>
<td>88.5</td>
<td>85.7</td>
</tr>
<tr>
<td>Crude protein (%)</td>
<td>15.2</td>
<td>18.9</td>
<td>16.9</td>
<td>14.6</td>
<td>21.5</td>
<td>17.1</td>
</tr>
</tbody>
</table>
Table 5. Aggregated diets per tonne of ration for four diets used for the national herd

<table>
<thead>
<tr>
<th>Ration component (kg)</th>
<th>NSW–Vic.</th>
<th>Qld</th>
<th>WA</th>
<th>SA</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>B</td>
<td>G-F</td>
<td>B</td>
<td>G-F</td>
</tr>
<tr>
<td>Barley (11%)</td>
<td>343.3</td>
<td>0.0</td>
<td>125.0</td>
<td>221.2</td>
</tr>
<tr>
<td>Sorghum (11%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>371.7</td>
</tr>
<tr>
<td>Wheat (12%)</td>
<td>517.0</td>
<td>825.1</td>
<td>737.3</td>
<td>231.6</td>
</tr>
<tr>
<td>Lupins (34%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Field peas (20.5%)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
</tr>
<tr>
<td>Bloodmeal (85%)</td>
<td>9.4</td>
<td>24.0</td>
<td>14.0</td>
<td>9.7</td>
</tr>
<tr>
<td>Meat and bone meal (50%)</td>
<td>25.4</td>
<td>30.5</td>
<td>25.0</td>
<td>18.4</td>
</tr>
<tr>
<td>Canola meal (34%)</td>
<td>37.1</td>
<td>50.0</td>
<td>66.0</td>
<td>88.3</td>
</tr>
<tr>
<td>Soymeal (48%)</td>
<td>20.2</td>
<td>18.0</td>
<td>10.0</td>
<td>10.2</td>
</tr>
<tr>
<td>Other protein meal</td>
<td>15.0</td>
<td>25.0</td>
<td>0.0</td>
<td>15.0</td>
</tr>
<tr>
<td>Vegetable oil</td>
<td>10.5</td>
<td>9.0</td>
<td>1.0</td>
<td>8.9</td>
</tr>
<tr>
<td>Low-cost additives</td>
<td>12.4</td>
<td>7.0</td>
<td>15.0</td>
<td>14.8</td>
</tr>
<tr>
<td>High-cost additives</td>
<td>10.1</td>
<td>11.4</td>
<td>6.8</td>
<td>10.2</td>
</tr>
<tr>
<td>Total (kg)</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
<td>1000</td>
</tr>
<tr>
<td>Diet digestibility (%)</td>
<td>80.8</td>
<td>84.3</td>
<td>83.8</td>
<td>80.3</td>
</tr>
<tr>
<td>Crude protein (%)</td>
<td>14.4</td>
<td>18.9</td>
<td>17.0</td>
<td>14.2</td>
</tr>
</tbody>
</table>

Major feed grains were modelled from Australian grain processes available from the AustLCI database (Life Cycle Strategies 2015) and Wiedemann et al. (2010). These processes were updated to include emission factors from the Australian NIR 2013 (Commonwealth of Australia 2015a). For major grains in each region, the proportion of grain produced in different systems (i.e. dry land and irrigated) was determined using the proportion of crop land irrigated in each state (ABS 2009, 2010, 2011). Using these data, grain processes were aggregated to provide an average market for the major grains in each state (see Table S1, available as supplementary material for this paper).

Where data were unavailable for some small dietary inputs such as vitamins, substitutions were made with other feed inputs, using product cost to guide the substitution. Feed-mill energy data were collected from three commercial feed-mills (see Table S1). These data were averaged and used for all CSFs and the national herd, with state-based electricity processes used.

Land-use and direct land use-change emissions

Land-use (LU) and dLUC emissions were assessed for Australian- and imported-crop production systems. The area sown annually to crops has expanded in Australia in the period 1990–2010 (Lesslie and Mewett 2013), and Wiedemann et al. (2016b) suggested that the expansion in land cropped was up to 21% in some states. An analysis of data from the Australian national inventory (Commonwealth of Australia 2015b) showed that annualised emissions associated with conversion of forest land to crop land were 4 755 000 t CO2-e in the period 1990–2010 (Commonwealth of Australia 2015c). The analysis of LU emissions from crop land were −4 800 000 t CO2-e (negative emissions indicate carbon sequestration), annualised over the same period. Carbon sequestration in Australian crop land is mostly in response to carbon sequestration resulting from adoption of improved cropping practices over the past 20 years. When divided by the average total land area sown to crops annually in Australia over the period 1990–2010, annualised emissions from LU and dLUC were −229 and 227 kg CO2-e/ha. Differences in LU emissions or sequestration are likely to exist among cropping regions in Australia based on specific management (Wiedemann et al. 2016b). However, in the present study, we accounted LU and dLUC emissions from crop land at the national scale, as suitable disaggregated datasets were not available to assess impacts associated with individual crops or cropping regions.

Imported soybean meal was modelled using data from the EcoInvent database (Swiss Centre for Life Cycle Inventories 2014) based on the relative imports of soybean meal from: Brazil (41%), Argentina (40%) and the USA (19%) (OEC 2015).

Table 6. Feed-milling energy inputs per tonne of ration delivered to the pig farm

<table>
<thead>
<tr>
<th>Input</th>
<th>Mean ± uncertainty</th>
</tr>
</thead>
<tbody>
<tr>
<td>Electricity (kWh/tᵃ)</td>
<td>32 ± 5.5</td>
</tr>
<tr>
<td>Diesel (L/t)</td>
<td>4.2 ± 2.3</td>
</tr>
<tr>
<td>Gas (MJ/t)</td>
<td>67 ± 49.7</td>
</tr>
<tr>
<td>Transport of commodities to feedmill (km)</td>
<td>85 ± 49.4</td>
</tr>
</tbody>
</table>

ᵃReported on an ‘as fed’ basis inclusive of moisture.
systems studied, spent litter was cleaned out after each batch of pigs, which is common practice in Australia.

Manure-management emissions were modelled from predicted manure excretion and feed waste using the PIGBAL manure-estimation model (Skerman et al. 2015) for each piggery and for the national herd. Briefly, the PIGBAL model uses a mass-balance approach to predict excreted nitrogen, and the dry-matter digestibility approximation of manure production method to determine excreted volatile solids. Feed waste is a predicted input to the manure stream. Manure emissions were determined using the emission factors outlined in the Australian NIR (Commonwealth of Australia 2015a). Key factors are shown in Table 7, and the mass of manure and waste feed relative to the functional unit is shown in Table 8. Integrated emission factors and manure mass flow data were used for the Australian national herd (Commonwealth of Australia 2015a) and these factors are provided in the supplementary material (Tables S2, S3). Uncertainty associated with emission factors was determined from the corresponding IPCC inventory methods (Dong et al. 2006).

**Meat processing**

Meat-processing inventory data were collected from four large pork-processing plants over a 12-month period. Data were averaged and used for all supply chains. Table 9 provides the input data to process 1000 kg of chilled pork.

### Table 7. Livestock greenhouse-gas parameters with uncertainty for case-study farms (CSFs)

<table>
<thead>
<tr>
<th>Emission source</th>
<th>Emission and units</th>
<th>Value</th>
<th>Uncertainty</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Manure – emissions from uncovered anaerobic pond</td>
<td>Ultimate methane yield (Bo)</td>
<td>0.45</td>
<td>±15%</td>
<td>(Commonwealth of Australia 2015a)</td>
</tr>
<tr>
<td>Methane-conversion factor (MCF)</td>
<td>0.75 (NSW)</td>
<td>±20%</td>
<td>(Commonwealth of Australia 2015a)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.77 (Qld)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.74 (Vic.)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.77 (WA)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from outdoor (dry lot)</td>
<td>0.01 (WA)</td>
<td>±20%</td>
<td>(Commonwealth of Australia 2015a)</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from deep litter</td>
<td>0.04 (NSW)</td>
<td>±20%</td>
<td>(Commonwealth of Australia 2015a)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.04 (WA)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from uncovered anaerobic pond</td>
<td>N₂O-N/kg N excreted</td>
<td>0</td>
<td>Not applicable</td>
<td>(Commonwealth of Australia 2015a), based on Dong et al. (2006)</td>
</tr>
<tr>
<td>Manure – emissions from outdoor (dry lot)</td>
<td>0.02</td>
<td>Factor of 2</td>
<td>(Commonwealth of Australia 2015a), based on Dong et al. (2006)</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from deep litter</td>
<td>0.01</td>
<td>Factor of 2</td>
<td>(Commonwealth of Australia 2015a), based on Dong et al. (2006)</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from uncovered anaerobic pond</td>
<td>NH₃-N/kg N excreted</td>
<td>0.55</td>
<td>±25%</td>
<td>(Commonwealth of Australia 2015a), based on Tucker et al. (2010) and Wiedemann et al. (2012a)</td>
</tr>
<tr>
<td>Manure – emissions from outdoor (dry lot)</td>
<td>0.3</td>
<td>±50%</td>
<td>(Commonwealth of Australia 2015a), based on Dong et al. (2006)</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from deep litter</td>
<td>0.125</td>
<td>±50%</td>
<td>(Commonwealth of Australia 2015a), based on Wiedemann et al. (2014)</td>
<td></td>
</tr>
<tr>
<td>Manure – emissions from stockpile</td>
<td>MCF</td>
<td>0.02 (NSW)</td>
<td>±20%</td>
<td>(Commonwealth of Australia 2015a)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.02 (WA)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>N₂O-N/kg N to stockpile</td>
<td>0.005</td>
<td>Factor of 2</td>
<td>(Commonwealth of Australia 2015a), based on Dong et al. (2006)</td>
<td></td>
</tr>
<tr>
<td>NH₃-N/kg N to stockpile</td>
<td>0.2</td>
<td>±25%</td>
<td>(Commonwealth of Australia 2015a), based on FSA Consulting (2007)</td>
<td></td>
</tr>
<tr>
<td>Indirect N₂O from volatilised NH₃</td>
<td>N₂O-N/kg N volatilised as NH₃</td>
<td>0.002</td>
<td>Factor of 2</td>
<td>(Commonwealth of Australia 2015a)</td>
</tr>
</tbody>
</table>

### Table 8. Manure output and feed-waste inventory

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Qld</th>
<th>SMC</th>
<th>Qld</th>
<th>LC</th>
<th>Vic.</th>
<th>LC</th>
<th>WA</th>
<th>LC</th>
<th>WA</th>
<th>O–DL</th>
<th>National herd</th>
</tr>
</thead>
<tbody>
<tr>
<td>Total manure VS (kg/100 kg LW)</td>
<td>48.9</td>
<td>40.7</td>
<td>36.5</td>
<td>38.4</td>
<td>41.3</td>
<td>48.1</td>
<td>51.8</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Total manure N (kg/100 kg LW) ²</td>
<td>6.4</td>
<td>6.3</td>
<td>5.4</td>
<td>6.4</td>
<td>4.8</td>
<td>6.0</td>
<td>5.9</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feed waste VS (kg/100 kg LW)</td>
<td>20.4</td>
<td>16.4</td>
<td>18.8</td>
<td>15.6</td>
<td>15.2</td>
<td>15.9</td>
<td>21.5</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feed waste (% of total feed fed)</td>
<td>7.5</td>
<td>6.5</td>
<td>5.5</td>
<td>7.7</td>
<td>6.7</td>
<td>5.8</td>
<td>7.6</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Feed waste (% of VS in waste stream)</td>
<td>43</td>
<td>40</td>
<td>36</td>
<td>42</td>
<td>37</td>
<td>33</td>
<td>41</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

¹This includes excreted VS and feed-waste VS.
²This includes excreted N and feed-waste N.
Allocation

Allocation processes are required for several points in the feed- and pig-production systems. In the feed-supply chain, economic allocation processes were used to determine impacts to protein meals and oil products (see supplementary material, Table S4). Where rendered products such as meat meal were included in the feed supply chain, only the impacts associated with rendering the product and transporting it were attributed to pig production.

The pig-supply chain delivers multiple products at different points in the system. Product yield and mass flow data were collected from abattoirs and detailed in Wiedemann and Yan (2014). LW from young slaughter pigs and older, culled breeding animals was aggregated to avoid allocation. Similarly in the meat-processing plant, all meat products entering the human supply chain, including edible offal, were aggregated to avoid allocation (Wiedemann and Yan 2014; LEAP 2014b, 2015). Manure and effluent were treated as residues after on-site treatment, as few piggeries generated sufficient revenue from manure or effluent sales to treat this as a co-product. Manure from conventional piggeries is typically land-applied on-site to crops, or pastures grazed by beef cattle or sheep. Solid residues such as sludge and spent litter are more readily transported off-site for application on crop land. Emissions arising from land application of these residues were allocated to the industry that utilised the manure nutrients. To account for the input of manure to crop systems, we assumed that 30% of manure nutrients were returned to the grain-production system, representing 0.6% of cereal crop fertilizer requirements nationally. Manure was included as an input to the modelled cereal crop systems used in the feed inventory.

At the meat-processing plant, pet food was produced as a co-product to meat. Economic value was used to determine the allocation of impacts to each product, with allocation fractions of 99.3% for meat and 0.7% for pet food. Renderable products were considered a residue, and no impacts were allocated to these products. The estimated edible fraction of the retail product was 99.3% for meat and 0.7% for pet food. Renderable products were treated as residues after on-site treatment, as few piggeries generated sufficient revenue from manure or effluent sales to treat this as a co-product. Manure from conventional piggeries is typically land-applied on-site to crops, or pastures grazed by beef cattle or sheep. Solid residues such as sludge and spent litter are more readily transported off-site for application on crop land. Emissions arising from land application of these residues were allocated to the industry that utilised the manure nutrients. To account for the input of manure to crop systems, we assumed that 30% of manure nutrients were returned to the grain-production system, representing 0.6% of cereal crop fertilizer requirements nationally. Manure was included as an input to the modelled cereal crop systems used in the feed inventory.

Scenario modelling

The impact of alternative housing and MMSs was investigated using two case-study piggeries. Scenario assumptions are described as follows:

Scenario 1 (S1): Qld LC – covered anaerobic pond (CAP) with combined heat and power (CHP) at the grower–finisher unit. The Qld LC piggery installed a CAP with CHP after the benchmarking period (2010–2011) and this scenario was modelled using site data collected in 2014. Under current operational conditions, all effluent from the grower–finisher pigs is treated in a CAP, or 55% of total manure from the piggery. Approximately 54% of the biogas is converted to electrical and heat energy in the CHP engine and the remainder is flared. Following treatment in the CAP, effluent flowed into a secondary pond. Scenario 2 (S2): Qld LC – CAP–CHP at the whole piggery. This scenario modelled the piggery operating at maximum potential for biogas production and energy recovery. It was assumed that all effluent produced at the piggery was treated in CAPs and all biogas produced was converted to electricity and heat in the CHP.

Scenario 3 (S3): Qld LC – DL. This scenario modelled a conversion of the animal houses to use deep litter for the grow-out and finishing stages. Manure emission factors for methane are significantly lower for deep-litter housing than for conventional effluent treatment (Commonwealth of Australia 2015a) and emissions per kilogram of pork produced are expected to be lower (Wiedemann et al. 2014). In addition, electricity consumption is lower for the -itter system, as it uses natural ventilation, whereas the Qld LC CSF is tunnel ventilated. A diagram of the CSF and scenario supply chain is provided in Fig. 2.

Scenario 4 (S4): Vic. LC – CAP. This piggery also installed a CAP following the benchmarking period, and data were collected to model the impact of this change as a scenario. This supply chain produces pigs on three different sites. The CAP was installed to treat effluent from all finisher pigs, and 56% of the grower pigs. In total, 70% of effluent is treated in the CAP and all biogas is flared.

Scenario 5 (S5): Vic. LC – CAP–CHP, all manure treated. This scenario modelled the piggery operating at maximum potential for biogas production, with biogas used to produce electricity and heat.

Scenario 6 (S6): Qld SMC – short hydraulic retention time (HRT) effluent storage and irrigation. This scenario modelled the Qld SMC average piggery with short-term storage and rapid irrigation to avoid emission generating anaerobic pond conditions. Modelling was done using assumptions from the Commonwealth of Australia (2015a). Assumptions for each scenario are provided in Table 10.

Modelling and statistical analyses

Modelling was undertaken using Simapro™ 8.0 (Pre-Consultants, Amersfoort, The Netherlands). Model uncertainty for the national herd was assessed using Monte Carlo analysis in SimaPro 8.0. One-thousand iterations provided a 95% confidence interval for the results. Multiple linear regression analysis was used to describe the influence of key herd parameters and farm variables on GHG emissions.

Results

Farm gate

Total GHG (excluding LU and dLUC) ranged from 2.1 to 4.5 kg CO₂-e/kg LW for the NSW C–DL, and Qld SMC respectively.
Emissions from the conventional CSFs (Qld SMC, Qld LC, Vic. LC and WA LC) were higher than for the alternative production systems (NSW C–DL and WA O–DL) in response to the very high emissions (averaging 64% of total GHG) from uncovered effluent treatment at the conventional piggeries. Nitrous oxide arose from both grain production and manure management, with the largest emissions observed from the NSW C–DL (25%) and WA O–DL (33%) piggeries where nitrous oxide emissions from the MMS were highest. Carbon dioxide associated with fossil-fuel use ranged from 23% to 42% of total emissions.

Total GHG emissions (excluding LU and dLUC) for the national herd was 3.6 kg CO₂-e/kg LW. The predominant factor influencing emissions was the proportion of pigs managed with high or low emission-intensity MMSs. A moderate proportion (35%) of the national herd is managed with deep litter or outdoor housing, or covered ponds. As a consequence, the national average emissions were 8% lower than the conventional CSFs. This difference may have been greater, except for the poorer FCR for the national herd than for the mean of the conventional CSFs.

Emissions from LU and dLUC sources ranged from 0.08 to 0.7 kg CO₂-e/kg LW for the WA LC and Vic. LC CSFs respectively. Losses predominantly arose from dLUC in Australia and South America (soymeal production), which was partially offset by carbon sequestration in Australian crop land. As LU and dLUC associated with Australian cropping largely cancelled each other, differences in net LU and dLUC emissions were associated with the rate of inclusion of soymeal in the diet and the FCR.

The linear regression analysis (Fig. 4) showed that the FCR was able to explain 0.88 of the variation (P < 0.001) in GHG intensity (excluding LU and dLUC) at conventional piggeries. The regression model of GHG emission intensity was

\[
GHG = 1.625FCR - 0.605(R^2 = 0.88).
\]

**Farm gate-scenario analysis**

Figure 5 shows the total GHG emissions for LW production for the Qld LC and Vic. LC CSFs (baseline), and the alternative housing and manure-management scenarios. For the Qld LC scenarios, emissions were reduced by 31%, 60% and 38% for the Qld LC – CAP (S1), Qld LC – CAP (S2) and Qld LC – DL (S3) respectively. For the Vic. LC scenarios, emissions were reduced by 44% and 64% for the Vic. LC – CAP (S4) and Vic. LC – CAP (S5) respectively. Emissions from the short HRT system were 57% lower than emissions from the Qld SMC with the current management, suggesting that this approach is also a significant mitigation opportunity.

Lower GHG and fossil fuel energy demand for the CAP–CHP scenario was the result of reduced methane emissions, reduced electricity demand at the piggeries and reduced gas use for heating at the piggeries and, therefore, provided more co-benefits than deep-litter housing or short-HRT systems.

**Wholesale pork – national herd**

Greenhouse-gas intensity (excluding LU, dLUC) from the national herd was 6.36 ± 1.03 kg CO₂-e/kg wholesale pork. The emission profile was dominated by methane from manure management, followed by ration production and then meat processing (Fig. 6). Emissions from LU and dLUC sources were 0.38 ± 0.5 kg CO₂-e/kg wholesale pork, with net emissions predominantly arising from dLUC associated with South American soymeal production.

**Discussion**

The current study presents the first comprehensive analysis of GHG emissions from Australian pork production by using LCA. Substantial differences were observed among housing and MMSs, and as a result of production efficiency, but fewer differences were observed among production regions. Conventional housing and open, anaerobic effluent-treatment systems generated high levels of manure emissions, contributing 2.2–3.0 kg CO₂-e/kg LW. The Australian National Inventory applies values that are close to the highest methane-conversion factors provided by the IPCC (Dong et al. 2006), because the very high temperatures and long-HRT systems used in Australia lead to high rates of methane production. Recent research (McGahan et al. 2016) has confirmed that emissions arising from these treatment systems

\[
\text{(Fig. 3). Emissions from the conventional CSFs (Qld SMC, Qld LC, Vic. LC and WA LC) were higher than for the alternative production systems (NSW C–DL and WA O–DL) in response to the very high emissions (averaging 64% of total GHG) from uncovered effluent treatment at the conventional piggeries. Nitrous oxide arose from both grain production and manure management, with the largest emissions observed from the NSW C–DL (25%) and WA O–DL (33%) piggeries where nitrous oxide emissions from the MMS were highest. Carbon dioxide associated with fossil-fuel use ranged from 23% to 42% of total emissions.}
\]
Table 10. Emission factors for alternative manure-management scenarios (S1–S6)

MMS, manure-management system; VS, volatile solids. S1, Qld LC – CAP–CHP at the grower–finisher unit; S2, Qld LC – CAP–CHP for the whole piggery; S3, Qld LC–DL for grow-out and finishing; S4, Vic. LC – CAP-flare with 70% of effluent treated in CAP; S5, Vic. LC – CAP–CHP for the whole piggery; S6, Qld SMC–SHRT for the whole piggery. See text for explanation of the codes.

<table>
<thead>
<tr>
<th>Parameter</th>
<th>S1 CAP–CHP</th>
<th>S2 CAP–CHP</th>
<th>S3 Deep litter</th>
<th>S4 CAP–flare</th>
<th>S5 CAP–CHP</th>
<th>S6 SHRT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Proportion of total waste treated in uncovered anaerobic pond (%)</td>
<td>45</td>
<td>0</td>
<td>24</td>
<td>30</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Proportion of total waste treated in alternative MMS (%)</td>
<td>55</td>
<td>100</td>
<td>76</td>
<td>70</td>
<td>100</td>
<td>100</td>
</tr>
<tr>
<td>Biogas yield (m³ biogas/ kg VS)</td>
<td>0.495</td>
<td>0.495</td>
<td>n.a</td>
<td>0.476</td>
<td>0.476</td>
<td>n.a</td>
</tr>
<tr>
<td>Proportion of methane in biogas (%)</td>
<td>70</td>
<td>70</td>
<td>n.a</td>
<td>70</td>
<td>70</td>
<td>n.a</td>
</tr>
<tr>
<td>Methane conversion factor (MCF)</td>
<td>n.a</td>
<td>n.a</td>
<td>0.04</td>
<td>n.a</td>
<td>n.a</td>
<td>0.03</td>
</tr>
<tr>
<td>Methane density (kg/m³)</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
</tr>
<tr>
<td>VS reduction (inc. partitioning to sludge) (%)</td>
<td>70</td>
<td>70</td>
<td>n.a</td>
<td>70</td>
<td>70</td>
<td>70</td>
</tr>
</tbody>
</table>

Secondary treatment and land application

<table>
<thead>
<tr>
<th>Secondary treatment system</th>
<th>Secondary pond</th>
<th>Secondary pond</th>
<th>Stockpile</th>
<th>Secondary pond</th>
<th>Secondary pond</th>
<th>Irrigation^</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nitrous oxide emissions (kg N₂O-N/ kg manure N)</td>
<td>0.0</td>
<td>0.0</td>
<td>0.005</td>
<td>0.0</td>
<td>0.0</td>
<td>0.01</td>
</tr>
<tr>
<td>Methane emissions (m³/kg VS)</td>
<td>0.092</td>
<td>0.092</td>
<td>0.009</td>
<td>0.089</td>
<td>0.089</td>
<td>0.00</td>
</tr>
<tr>
<td>Methane density (kg/m³)</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>0.668</td>
<td>n.a</td>
</tr>
<tr>
<td>VS reduction (inc. partitioning to sludge) (%)</td>
<td>40</td>
<td>40</td>
<td>n.a</td>
<td>40</td>
<td>40</td>
<td>n.a</td>
</tr>
</tbody>
</table>

Biogas utilisation system

| Proportion of total biogas used in CHP to produce energy (remainder is flared) (%) | 54 | 100 | n.a | 0 | 100 | n.a |
| Electrical efficiency CHP (%) | 30 | 30 | n.a | 30 | 30 | n.a |
| Thermal efficiency CHP (%) | 45 | 45 | n.a | 45 | 45 | n.a |

^Emissions and fertiliser value of irrigated manure was attributed to the receiving crop.

Fig. 3. Total greenhouse-gas emissions (excluding land use (LU) and direct land-use change (dLUC)) per kilogram of liveweight for six case-study supply chains and the Australian national herd.
are very high by global standards. As a result of the higher emissions from manure management, emissions from conventional pork-production systems in Australia tended to be higher than emissions from northern hemisphere countries (i.e. Pelletier et al. 2010; Reckmann et al. 2012).

We found that deep-litter housing for grower and finisher pigs produced ~30% lower emissions than did conventional housing of grower and finisher pigs (Scenario 3) in response to lower emissions from manure management. Total emissions per kilogram LW for the deep-litter systems were similar to those in other studies (Basset-Mens and van der Werf 2005; Pelletier et al. 2010). Manure excreted in a deep-litter house is managed in a predominantly aerobic environment, leading to low methane emissions. While nitrous oxide emissions are higher from manure managed in deep-litter sheds than from that in conventional sheds, net GHG emissions were considerably lower, suggesting that this housing system offers a mitigation opportunity (Phillips et al. 2016). GHG emissions from outdoor housing were also lower than those from conventional production in the present study. We note that the study relied on default emission factors from manure excreted in outdoor pig-farming areas and this may have increased uncertainty in the result. However, the nitrous oxide factor applied to excreted manure is ~10 times the equivalent Australian factor for nitrous oxide emitted from synthetic fertiliser, and double the factor for land-applied manure emissions. Thus, it is likely to be conservative. While the outdoor CSF had poorer production efficiency (pigs weaned per sow per year and FCR), which resulted in higher feed-
related impacts, manure emissions were considerably lower for reasons similar to those in the deep-litter housing. While GHG impacts were lower, other impacts such as eutrophication may be a concern with outdoor pig farming, as a result of off-site nutrient export and leaching of nutrients, and would need to be assessed to provide a more rounded environmental assessment.

The scenario modelling showed large reductions in GHG emissions for conventional housing systems where CAP and CHP or short-HRT systems were installed to replace open effluent ponds. Two scenarios (Scenarios 1 and 4) were based on actual installed systems at the case-study piggeries and, therefore, provided insight into the likely emission reductions under commercial conditions. These piggeries housed breeder, weaner and grower–finisher pigs in different houses at distances of up to 8 km apart. The CAP systems were installed at the grower–finisher sites, where the largest amount of manure is produced. At the Qld LC site, the heat produced by the CHP was not fully utilised, because it was logistically difficult to transport heat from the grower–finisher site to the breeder site, where most heat is required. This is a common problem for multi-site piggeries and may limit the theoretical capability of CHP units to meet the heat requirements of the piggery. We observed that transporting electricity from the generation site to other sites was logistically difficult, requiring, in some instances, the installation of privately owned power networks with high capital costs. We also observed that excess electricity was not easy to sell by exporting to the grid at the Qld LC site, and, at the time of performing the analysis, cost-effective agreements had not been established with local power providers. As a result, excess biogas was flared to destroy the methane, but the energy potential was not utilised.

Where we modelled all manure being treated in CAP–CHP systems (Scenarios 2 and 5), total emissions at the farm level were reduced by 60% and 64%, to 1.6 and 1.4 kg CO$_2$-e/kg LW. When converted to a carcass-weight basis, the lowest impacts for pork are in the order of 2.3 kg CO$_2$-e/kg carcass weight, which is approaching a level of efficiency similar to that for chicken meat produced in Queensland, Australia (Wiedemann et al. 2012b).

As an alternative to CAP–CHP systems, we modelled a simple short-HRT system where anaerobic treatment is avoided through rapid irrigation of effluent to land. This system may be more cost effective at small piggeries, where the high capital cost of installing CAP systems is prohibitive. This also provided substantial reductions in GHG emissions (57%). However, we note that short-HRT systems, which are expected to have <30 days of effluent storage, may be more difficult to manage to ensure a minimal risk of nutrient loss when irrigating, when prolonged wet weather periods are experienced. This may be addressed by maintaining additional wet-weather storage systems, but diverting effluent away from these systems to direct irrigation whenever conditions permit this practice, and further research and development are required to develop practical and sustainable systems that meet all environmental objectives.

According to the National Inventory, 6.5% of manure was treated in CAPs or engineered digesters in the 2010–2013 period (derived from Commonwealth of Australia 2015a, and Wiedemann et al. 2014). Considering a large proportion of manure is currently treated in anaerobic effluent systems (see Tables S3, S4 in the Supplementary material), the total mitigation potential for the industry is substantial, provided sufficient incentives exist for commercial producers to install CAPs or engineered digesters. On the basis of the current contribution of emissions from manure sources for the national average, a further 30% reduction in emissions for the cradle-to-farm gate may be achievable, assuming installation of CAPs or digesters reached 50% of the industry.
Feed production contributed 27% of total emissions (excluding LU and dLUC) for the national herd. Total feed impacts (excluding LU and dLUC) ranged from 304 to 366 kg CO$_2$e/tonne feed fed and slightly higher when LU and dLUC emissions were included. Impacts from feed production were primarily associated with energy use in field operations, and field emissions of nitrous oxide. The variation in diet impacts from the most emission intensive diet to the least emission intensive suggests that some opportunity exists to reduce impacts by selecting lower emission intensity diets. Emissions from LU and dLUC sources were smaller than what might have been expected, because while dLUC emissions from the expansion of Australian crop land were substantial, carbon sequestration in established Australian crop land is also substantial (Commonwealth of Australia 2015a) and these two factors counter-act each other.

Impacts associated with feed production are more difficult to manage for the pork industry. Feed formulations are prepared on a least-cost basis, and selecting preferred ingredients to reduce environmental impacts may have cost implications, making it a less attractive strategy for mitigation. However, improving FCR will reduce feed-related impacts, and will also reduce manure production and, therefore, manure-related emissions. Reducing FCR may also reduce cost-of-production and most Australian farmers focus strongly on this. The strong relationship shown between FCR and GHG emissions in the present study confirmed that focussing on this aspect of production can result in both environmental and production benefits for the industry.

When impacts were assessed through to the point of wholesale distribution for the national herd, emissions were dominated by primary production, manure management, feed production and, to a lesser extent, energy use. Meat processing contributed a smaller proportion of impacts. However, because of the mass losses associated with meat processing, reported impacts per kilogram of pork rose substantially. Impacts per kilogram of edible product (edible yield estimated at 85%) were 6.5 kg CO$_2$e, or 7.4 kg CO$_2$e/kg edible product, with LU and dLUC emissions included. The contribution from LU and dLUC emissions was 6% of total emissions. Impacts from Australian pork production tended to be higher than results from European pork production (Basset-Mens and van der Werf 2005; Nguyen et al. 2011; Reckmann et al. 2012), which were in the order of 3.7–4.0 kg CO$_2$e/kg pork when converted to a functional unit equivalent to that in the present study, accounting for mass losses with further meat processing. The higher emissions from Australian production corresponded to higher manure-management emissions. Emissions associated with LU and dLUC were not reported by these authors, although the authors indicated that soymeal from South America is fed and, therefore, emissions would be expected from this feed source.

Emissions from pork production were found to be lower than from Australian lamb or beef (Wiedemann et al. 2015b), but higher than from Australian chicken meat (Wiedemann et al. 2012b), noting that the latter did not include impacts from LU and dLUC. Considering the large reduction in emissions possible by managing manure emissions in pork production and the ongoing adoption of this technology in the industry, it is expected that emissions from Australian pork will decline over time.

Conclusions

The present study is the first case study and national analysis of GHG emissions from Australian pork production by using LCA. Emissions from primary production were found to dominate GHG through to production of a wholesale product. In conventional housing systems, emissions were found to be strongly related to FCR, and improvement in FCR across the industry is expected to result in lower emissions over time. The emission profile was strongly influenced by housing and MMS type, and opportunities exist to reduce emissions from this source. Emissions were found to be lower from alternative housing systems such as deep litter, suggesting that these systems may provide mitigation opportunities to the industry. Anaerobic digestion and electricity generation from effluent may substantially reduce emissions from conventional piggeries, and adoption of this technology is expected to reduce emissions from the national herd over time.

Acknowledgements

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References


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9.2 PAPER 2. ENERGY, WATER, LAND OCCUPATION

9.2.1 ATTRIBUTION STATEMENT

The paper *Environmental impacts and resource use from Australian pork production determined using life cycle assessment: 2. Energy, Water and Land Occupation* was led by S.G Wiedemann and co-authored by E.J McGahan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
- Development and application of the greenhouse gas model
- Development of the water use model
- Development and application of grain processes used in the modelling of feed inputs
- Data acquisition and data analysis, including impact assessment
- Allocation and uncertainty analysis
- Preparation of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model and herd modelling

Murphy contributed to the study and manuscript via:

- Assistance with data analysis and assistance with preparation of figures, tables and manuscript
Environmental impacts and resource use from Australian pork production determined using life cycle assessment.  
2. Energy, water and land occupation

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\textsuperscript{B}2 Somerset Street, Toowoomba, QLD 4350, Australia.  
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\textsuperscript{D}Corresponding author. Email: stephen.wiedemann@integrityag.net.au

Abstract. Utilisation of water, energy and land resources is under pressure globally because of increased demand for food, fibre and fuel production. Australian pork production utilises these resources both directly to grow and process pigs, and indirectly via the consumption of feed and other inputs. With increasing demand and higher costs associated with these resources, supply chain efficiency is a growing priority for the industry. This study aimed to quantify fresh water consumption, stress-weighted water use, fossil fuel energy use and land occupation from six case study supply chains and the national herd using a life cycle assessment approach. Two functional units were used: 1 kg of pork liveweight (LW) at the farm-gate, and 1 kg of wholesale pork (chilled, bone-in). At the farm-gate, fresh water consumption from the case study supply chains ranged from 22.2 to 156.7 L/kg LW, with a national average value of 107.5 L/kg LW. Stress-weighted water use ranged from 6.6 to 167.5 L H$_2$O-e/kg LW, with a national average value of 103.2 L H$_2$O-e/kg LW. Fossil fuel energy demand ranged from 12.9 to 17.4 MJ/kg LW, with a national average value of 14.5 MJ/kg LW, and land occupation ranged from 10.9 to 16.1 m$^2$/kg LW, with a national average value of 16.1 m$^2$/kg LW and with arable land representing 97% to 99% of total land occupation. National average impacts associated with production of wholesale pork, including impacts from meat processing, were 184 ± 43 L fresh water consumption, 172 ± 53 L H$_2$O-e stress-weighted water, 27 ± 2.6 MJ fossil fuel energy demand and 25.9 ± 5.5 m$^2$ land/kg wholesale pork. Across all categories through to the wholesale product, resource use was highest from the production of feed inputs, indicating that improving feed conversion ratio is the most important production metric for reducing the resource use. Housing type and energy generation from manure management also influence resource use requirements and may offer improvement opportunities.

Additional keywords: agricultural systems LCA, water footprint.

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Introduction

Future requirements for food, fibre and fuel production will place increased pressure on the global natural resource base to produce more from less. Globally, meat demand is expected to increase 74% by 2050 because of expanding global population and increased wealth (FAO 2009). However, global arable land and available water resources are constrained (FAO 2009) and fossil fuel resources are finite. In Australia, arable land resources are limited by soil type, climate and vegetation regulations to ~4% of national land mass (Lesslie and Mewett 2013). Water resources in Australia’s most heavily populated and water-stressed river basin, the Murray–Darling, are capped, and this restricted supply has led to increased competition between water users (MDBA 2012). For the pork industry, improving resource efficiency is an important strategy for maintaining access to a shrinking pool of resources without dramatically increasing costs. Presently, few data are available to allow the industry to benchmark performance and measure future improvements. Comprehensive assessment methods such as life cycle assessment (LCA) quantify impacts throughout a whole supply chain, including those associated with inputs such as feed. Such tools are effective in comparing different management systems because impacts from the whole system are taken into account, and impacts are reported relative to production. LCA has been applied in Australia to determine supply chain water use, energy use and land occupation using individual farm case studies (i.e. Peters et al. 2010; Eady et al. 2011; Wiedemann and McGahan 2011), and at the regional or national scale for beef (Wiedemann et al. 2015\textsuperscript{a}, 2016\textsuperscript{b}), export lamb (Ridoutt et al. 2012; Wiedemann et al. 2015\textsuperscript{c}, 2016\textsuperscript{d}) and chicken meat (Wiedemann et al. 2017). This study provides the first case study and national analysis of water, energy and land associated with Australian pork production using LCA, and is a companion study to the greenhouse gas LCA of Wiedemann et al. (2016\textsuperscript{a}). The study aimed to benchmark resource use, determine impact hotspots throughout the supply chain, and quantify the reduced impacts from several improvement strategies that may be applied on Australian farms.
Materials and methods

Goal and scope

The study was an attributional investigation of pork production from major production regions and different production systems in Australia, to provide information to the pork industry, research community and the general public.

Fossil fuel energy demand was assessed by aggregating all fossil fuel energy inputs throughout the system and reporting these per mega joule (MJ) of energy, using Lower Heating Values. Fresh water consumption was assessed using methods consistent with ISO 2014, described in the following sections. Assessment of stress-weighted water use was based on Pfister et al. 2009, and values were divided by the global water stress index (WSI) 0.602 and expressed as a water equivalent (H2O-e, Ridoutt and Pfister 2010). Land occupation was assessed by aggregating impacts throughout the supply chain, and both total land occupation and arable land occupation is reported in square meter years (m² year).

The primary production supply chain including breeding through to finishing (sometimes at multiple sites) and meat processing was included, with all associated inputs. Data were collected and impacts were assessed from 14 case study farms (CSF), grouped into six supply chains. The supply chains are described by state, piggery size and housing type, as follows: Qld small – medium conventional (Qld SMC), Qld large conventional (Qld LC), NSW conventional housing (breeding pigs) and deep litter housing for grower-finisher pigs (NSW C-DL), Victorian large conventional (Vic. LC), Western Australian large conventional (WA LC) and WA Outdoor housing (breeder pigs) and deep litter housing for grower-finisher pigs (WA O-DL). A national assessment was performed using national survey statistics for the year 2010. Descriptions of data collection methods are provided in Wiedemann et al. 2016a and herd performance data are reproduced in Table 1. The end-point of the supply chain was the cold storage unit where pork is stored before wholesale distribution. Results are presented using two functional units (FU): 1 kg of pork liveweight (LW) at the farm-gate, and 1 kg of chilled, bone-in wholesale pork cuts. The system boundary of the study is shown in Fig. 1 with the dashed line denoting the foreground system. The red arrow represents the flow of gilts (young females) and boars back into the breeding herd.

Table 1. Case study farm (CSF) herd production data based on primary data from major production regions

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C-DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O-DL</th>
<th>National herd (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Litters/sow.year</td>
<td>2.1</td>
<td>2.4</td>
<td>2.3</td>
<td>2.3</td>
<td>2.3</td>
<td>2.1</td>
<td>2.3 ± 3.5</td>
</tr>
<tr>
<td>Pigs weaned/sow.year</td>
<td>19.1</td>
<td>22.4</td>
<td>20.0</td>
<td>24.7</td>
<td>24.7</td>
<td>16.5</td>
<td>21.4 ± 9.4</td>
</tr>
<tr>
<td>No. of sows</td>
<td>318</td>
<td>8229</td>
<td>4680</td>
<td>1950</td>
<td>6370</td>
<td>1956</td>
<td>231 647 ± 4.0</td>
</tr>
<tr>
<td>Sow mortality rate (%)</td>
<td>6%</td>
<td>9%</td>
<td>5%</td>
<td>5%</td>
<td>10%</td>
<td>10%</td>
<td>10 ± 1.8</td>
</tr>
<tr>
<td>Sow culling rate (%)</td>
<td>33%</td>
<td>32%</td>
<td>46%</td>
<td>36%</td>
<td>31%</td>
<td>41%</td>
<td>37 ± 4.0</td>
</tr>
<tr>
<td>Average sale weight of slaughter pigs (kg)</td>
<td>104.1</td>
<td>84.2</td>
<td>99</td>
<td>107</td>
<td>96.7</td>
<td>102.3</td>
<td>97.4 ± 5.1</td>
</tr>
<tr>
<td>Average age of slaughter pigs (days)</td>
<td>155.2</td>
<td>136.5</td>
<td>156</td>
<td>154</td>
<td>160.2</td>
<td>176.3</td>
<td>151.4 ± 5.5</td>
</tr>
<tr>
<td>FCR (whole herd)</td>
<td>3.2</td>
<td>2.9</td>
<td>2.7</td>
<td>2.4</td>
<td>2.7</td>
<td>3.2</td>
<td>3.1 ± 6.7</td>
</tr>
</tbody>
</table>

*Data were averaged independently across the five piggeries.

**Weighted average of all pigs sold.**
of crop land irrigated in each state (ABS 2009, 2010, 2011). Losses associated with irrigation water supply of 27.1% were applied, based on the ABS national water accounts (ABS 2012). Grain processes were aggregated to provide an average market for the major grains in each state (see Wiedemann et al. 2016a; Supplementary material). The area of land occupation associated with crop production was a function of crop yield (see Supplementary material), and calculated in each grain inventory process. Crop yields were cross checked against national datasets to ensure representativeness, and further details of the crop inventory are provided in Wiedemann et al. (2016a).

Feed mill energy and water data were collected from three commercial feed mills. Water use averaged 113 L/tonne of ration produced and energy data were detailed in Wiedemann et al. (2016a). These data were averaged and used for all CSF and the national herd, with state-based electricity processes used.

**Fresh water consumption**

Water inputs were measured using water meters at most piggeries. A water balance model was established for each piggery to determine fresh water flows and water consumption throughout the piggery. Additionally, water balances were used to determine flows and losses in the water supply system. Within the piggery, water is used for livestock drinking, cleaning and cooling, and outputs occur via evaporation, respiration, incorporation into product and flows into the manure management system. Drinking water was either incorporated into the product and exported from the catchment (consumptive use) or voided in urine or manure. Manure and urine, together with cleaning water and spilled water in conventional systems, enters the effluent treatment system where water is returned to the atmosphere either by direct evaporation from effluent ponds, or via evapo-transpiration of irrigated effluent on nearby pastures. In sheds using deep litter, the moisture is captured in the litter, and is assumed to evaporate from the litter during storage or during land application. Thus, all water inputs were consumed by the system via multiple pathways and detailed water balance processes for the animal sheds could be simplified by treating all water inputs as consumptive uses, though the pathways were diverse.

**Fig. 1.** System boundary with foreground system boundary noted within the dashed lines.
We note that where effluent water was used as an input to cropping systems, this water could be attributed to the crop system rather than the pig system, noting the quality change that had occurred (Bayart et al. 2010). However, in the present study, beneficial effluent water reuse was not common and a conservative approach was applied by attributing all water consumption from effluent irrigation to the piggery system. For the purposes of reporting, freshwater consumption at the piggery was accounted for at the point where water evaporated from the system, which occurred either from the water supply system, directly from the sheds (including respiration losses from animals and evaporation of spilled drinking water, cleaning water, cooling water and excreted manure and urine), from the manure management system, or from the field after effluent or manure is applied to land. Irrigation water use associated with purchased commodities is described as ‘ration irrigation off-site’. Irrigation water, used to produce purchased feed inputs, was modelled using methods described in Wiedemann et al. (2015a). Aggregated water use inventory data including details of the water supply system and efficiency are presented in Table 3.

### Stress-weighted water use

The stress-weighted water use impact assessment method applied different stress weighting factors for different regions of Australia where the piggeries and meat processing plants were located, based on Pfister et al. (2009). For background products that may be sourced from many regions, we applied the Australian average WSI value of 0.402 for these sources. To calculate the stress-weighted water use, freshwater consumption in each region was multiplied by the relevant WSI and summed across the supply chain. The value was then divided by the global average WSI (0.602) and was expressed as water equivalents (H$_2$O-e; Ridoutt and Pfister 2010). Regional WSI values are shown in Table 3, and show two piggeries falling in regions of high water stress (WSI = 0.81–0.85) in the Murray–Darling basin of south-eastern Australia. Piggeries located in south-east Queensland and WA were in lower water stressed regions, though it should be noted that the WSI values were generated at a coarse level of resolution and some caution should be applied in their interpretation.

#### Meat processing

Meat processing inventory data were collected from four large pork processing plants over a 12-month period. Data were averaged and used for all supply chains as reported in Wiedemann et al. (2016a). Water inputs were 8050 L/1000 kg chilled pork and land areas for the meat processing plant were negligible because of the very high throughput of the plants and small land footprint. Packaging associated with the wholesale product was excluded.

#### Allocation

Allocation processes are required for several points in the feed and pig production systems and methods are described thoroughly in Wiedemann et al. (2016a), with a brief description provided here. In the feed supply chain, economic allocation processes were used to determine impacts to protein meals and oil products. Where rendered products such as meat meal were included in the feed supply chain, only the impacts associated with rendering the product and transporting it were attributed to pig production. Liveweight from young slaughter pigs and older, culled breeding animals was aggregated to avoid allocation. Manure and effluent were treated as residues after on-site treatment and no impact from pork production was attributed to this co-product. Meat yield was inclusive of edible offal, and impacts to co-products from meat processing were

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### Table 2. Aggregated general services and energy inputs for case study farms (CSF) and national herd per 100 kg of finisher liveweight (LW) sold

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C-DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O-DL</th>
<th>National herd (mean ± uncertainty)$^A$</th>
</tr>
</thead>
<tbody>
<tr>
<td>Purchased feed (kg/100 kg LW)</td>
<td>324.6</td>
<td>289.9</td>
<td>264.9</td>
<td>240.8</td>
<td>263.0</td>
<td>321.9</td>
<td>313.5 ± 26.7</td>
</tr>
<tr>
<td>Straw$^B$ (kg/100 kg LW)</td>
<td>n.a.</td>
<td>n.a.</td>
<td>23.1</td>
<td>n.a.</td>
<td>n.a.</td>
<td>69.9</td>
<td>18.9 ± 6.2</td>
</tr>
<tr>
<td>Diesel (L/100 kg LW)</td>
<td>0.56</td>
<td>0.51</td>
<td>0.23</td>
<td>0.79</td>
<td>0.32</td>
<td>1.10</td>
<td>0.41 ± 0.29</td>
</tr>
<tr>
<td>Petrol (L/100 kg LW)</td>
<td>0.37</td>
<td>0.22</td>
<td>0.10</td>
<td>0.11</td>
<td>1.25</td>
<td>0.16</td>
<td>0.1 ± 0.08</td>
</tr>
<tr>
<td>LPG (L/100 kg LW)</td>
<td>0.27</td>
<td>0.28</td>
<td>1.94</td>
<td>0.17</td>
<td>0.17</td>
<td>0.35</td>
<td>0.2 ± 0.19</td>
</tr>
<tr>
<td>Electricity (kWh/100 kg LW)</td>
<td>15.45</td>
<td>22.21</td>
<td>16.83</td>
<td>22.03</td>
<td>20.97</td>
<td>2.95</td>
<td>16.0 (6.3–26.5)$^C$</td>
</tr>
<tr>
<td>Accounting, auditing and bookkeeping (AUS/100 kg LW)</td>
<td>1.5</td>
<td>1.6</td>
<td>1.4</td>
<td>1.7</td>
<td>1.7</td>
<td>2</td>
<td>1.7 ± 0.09</td>
</tr>
<tr>
<td>Automotive repairs (AUS/100 kg LW)</td>
<td>2.7</td>
<td>2.7</td>
<td>2.5</td>
<td>3.3</td>
<td>3</td>
<td>8.3</td>
<td>3.3 ± 1.04</td>
</tr>
<tr>
<td>Veterinary products and services (AUS/100 kg LW)</td>
<td>6.4</td>
<td>7.6</td>
<td>7</td>
<td>8.3</td>
<td>8.5</td>
<td>12.4</td>
<td>8.3 ± 0.89</td>
</tr>
</tbody>
</table>

$^A$The uncertainty is reported as the 95% confidence interval, based on the mean, standard deviation and standard error. Values were assumed to follow a normal distribution.

$^B$Deep-litter pigs only.

$^C$Range in electricity values produced a positively skewed distribution, meaning that the s.d. gives no information on the asymmetry. Hence, the first and third quartiles were used as the upper and lower bounds of the range. Values were assumed to follow a triangular distribution.
Table 3. Piggery water resources for six case study supply chains and the Australian national herd

<table>
<thead>
<tr>
<th></th>
<th>Qld SMC</th>
<th>Qld LC</th>
<th>NSW C-DL</th>
<th>Vic. LC</th>
<th>WA LC</th>
<th>WA O-DL</th>
<th>National herd</th>
</tr>
</thead>
<tbody>
<tr>
<td>Average annual rainfall (mm)&lt;sup&gt;a&lt;/sup&gt;</td>
<td>650</td>
<td>650</td>
<td>589</td>
<td>369</td>
<td>650</td>
<td>530</td>
<td>n.a.</td>
</tr>
<tr>
<td>Piggery water supply</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dam (%)</td>
<td>0%</td>
<td>17%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>27%</td>
<td>5%</td>
</tr>
<tr>
<td>Bore (%)</td>
<td>100%</td>
<td>83%</td>
<td>100%</td>
<td>100%</td>
<td>100%</td>
<td>73%</td>
<td>85%</td>
</tr>
<tr>
<td>River/Creek (%)</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>5%</td>
</tr>
<tr>
<td>Reticulated (%)</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>0%</td>
<td>5%</td>
</tr>
<tr>
<td>Dam efficiency factor</td>
<td>n.a.</td>
<td>0.07</td>
<td>n.a.</td>
<td>n.a.</td>
<td>n.a.</td>
<td>0.5</td>
<td>n.a.</td>
</tr>
<tr>
<td>Bore efficiency factor</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
<td>1.00</td>
</tr>
<tr>
<td>Total water supply (ML)</td>
<td>24.1</td>
<td>367.5</td>
<td>35.0</td>
<td>100.5</td>
<td>230</td>
<td>63.1</td>
<td>10 403</td>
</tr>
<tr>
<td>Water stress index values</td>
<td>0.017</td>
<td>0.021</td>
<td>0.815</td>
<td>0.8545</td>
<td>0.011</td>
<td>0.01</td>
<td>0.402</td>
</tr>
</tbody>
</table>

<sup>a</sup>Recorded for nearest major town.

allocated using economic value, resulting in a very small allocation (0.7%) to co-products, based on reported market values in 2014.

Scenario modelling

The impact of alternative housing and manure management systems on energy demand and fresh water consumption was investigated using a series of scenarios at one case study piggery. Scenario assumptions are described as follows.

Scenario 1 (S1): Qld LC – covered anaerobic pond (CAP) with combined heat and power (CHP) at the grower-finisher unit. The Qld LC piggery installed a CAP with CHP after the benchmarking period (2010–11) and this scenario was modelled using site data collected in 2014. Under current operational conditions, all effluent from the grower/finisher pigs is treated in a CAP, or 55% of total manure from the piggery. Approximately 54% of the biogas is converted to electrical and heat energy in the CHP engine and the remainder is flared. Following treatment in the CAP, effluent flowed into a secondary pond.

Scenario 2 (S2): Qld LC – CAP with CHP at the whole piggery. This scenario modelled the piggery operating at maximum potential for biogas production and energy recovery. It was assumed that all effluent produced at the piggery was treated in the CAP and all biogas produced was converted to electricity and heat in the CHP.

Scenario 3 (S3): Qld LC – DL. This scenario modelled a conversion of the sheds to use deep litter for the grow-out and finishing stages. Electricity consumption is lower for the deep litter system, as it uses natural ventilation, whereas the Qld LC CSF is tunnel ventilated. In addition, fresh water consumption will be lower as less is required for removing manure from the sheds and cleaning. Assumptions for each scenario are provided in Table 4.

Modelling and uncertainty analysis

Modelling was carried out using Simapro™ 8.0 (PRé-Consultants 2014). An uncertainty analysis was carried out to establish the robustness of the national herd results for fresh water consumption, stress-weighted water use, fossil fuel energy demand and land occupation (see Supplementary material). Model uncertainty for the national herd was assessed using Monte Carlo analysis in SimaPro 8.0. One-thousand iterations provided a 95% confidence interval for the results.

Table 4. Scenario modelling parameters

<table>
<thead>
<tr>
<th>Scenario</th>
<th>S1 CAP-CHP</th>
<th>S2 CAP-CHP</th>
<th>S3 Deep litter</th>
</tr>
</thead>
<tbody>
<tr>
<td>Proportion of total waste treated in uncovered anaerobic pond</td>
<td>45%</td>
<td>55%</td>
<td>45%</td>
</tr>
<tr>
<td>Proportion of total waste treated in alternative MMS</td>
<td>100%</td>
<td>100%</td>
<td>65%</td>
</tr>
<tr>
<td>Total water supply (ML)</td>
<td>367.5</td>
<td>367.5</td>
<td>262.8</td>
</tr>
<tr>
<td>Water supply (L/kg LW)</td>
<td>21.9</td>
<td>21.9</td>
<td>15.7</td>
</tr>
<tr>
<td>Biogas utilisation system</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Biogas yield (m³ biogas/kg VS)</td>
<td>0.495</td>
<td>0.495</td>
<td>n.a</td>
</tr>
<tr>
<td>Proportion of methane in biogas</td>
<td>70%</td>
<td>70%</td>
<td>n.a</td>
</tr>
<tr>
<td>Methane density (kg/m³)</td>
<td>0.668</td>
<td>0.668</td>
<td>n.a</td>
</tr>
<tr>
<td>Proportion of total biogas used in CHP to produce energy (remainder is flared)</td>
<td>54%</td>
<td>100%</td>
<td>n.a</td>
</tr>
<tr>
<td>Electrical efficiency CHP</td>
<td>30%</td>
<td>30%</td>
<td>n.a</td>
</tr>
<tr>
<td>Thermal efficiency CHP</td>
<td>45%</td>
<td>45%</td>
<td>n.a</td>
</tr>
</tbody>
</table>

Results

Farm-gate

Fresh water consumption ranged from 22.2 to 156.7 L/kg LW at the farm-gate, for the WA LC and Qld LC supply chains respectively (Fig. 2). Fresh water consumption was dominated by irrigation water associated with feed grain production for CSF in the eastern states, ranging from 54% to 85%, but not in WA where there is less irrigated feed available. Water supply losses, respiration, and cooling water evaporative losses ranged from 3% to 36% for the conventional piggeries, with the largest differences being between naturally ventilated piggeries (i.e. the Qld SMC) and piggeries with evaporative cooling (Qld LC, WA LC) where water use was higher. The Qld LC piggery also had higher water use because of evaporative losses from water supply dams located on the farm, which contributed some 32% of total water use at this farm.

Supply losses, respiration and cooling was a larger proportion of water use for the outdoor piggery (64%) because this piggery had high supply losses from open storages, and high cooling water use for wallows.
Fresh water consumption (evaporation) losses from manure management systems ranged from 7.1% (Qld LC) to 35.6% (WA LC), and losses from land application were 2–20% from the conventional or deep litter piggeries. With similar diets, water use tended to be lower from deep litter compared with conventional finishing systems because the former required less water for cleaning. Total fresh water consumption at the WA O-DL supply chain (25.6 L/kg LW) was similar to the WA LC supply chains (22.2 L/kg LW) despite the very different production systems. The conventional piggery used larger amounts of water for cleaning and had high levels of productivity from the pigs. The outdoor piggery used large amounts of water for cooling (wallows) and losses from the water supply system were high, whereas cleaning water requirements were very low and herd performance was relatively lower, thus resulting in higher water use than may be expected.

Total fresh water consumption for the national herd was 107.5 ± 28.4 L/kg LW. The majority of water was associated with irrigated cereal grains produced in Australia and to a small extent, imported soy meal. Smaller amounts of water were consumed directly in pig production and processing.

Stress-weighted water use ranged from 6.6 to 167.5 L H$_2$O-e/kg LW, with the lowest values associated with the WA piggeries and the highest values from the NSW and Victorian piggeries. Stress-weighted water use was predominantly influenced by the region from which water was drawn, and to a lesser extent the volume of water used. The national average was 103.2 ± 32.4 L H$_2$O-e/kg LW.

Mean fossil fuel energy demand ranged between 12.9 and 17.4 MJ/kg LW (see Table 5 and Fig. 3). Feed production was the largest contributor to energy demand, ranging from 59% to 72%, followed by piggery energy use (28–41%). Among the conventional CSF, on-farm energy demand varied by 23% between the lowest and highest values, and total energy demand varied 26%. Deep litter and outdoor CSF were not found to be substantially different to conventional production with respect to energy demand, because the dominant impacts from feed production were similar in both housing systems and because of case study-specific aspects with these systems. Total fossil fuel energy demand for the national herd was 14.5 ± 1.1 MJ/kg LW, with 73% of impacts arising from feed production and the remainder from on-site energy use at piggeries.

Arable land occupation was lowest for the Qld LC supply chain at 10.9 m$^2$/kg LW and highest for the WA O-DL supply chain at 16.1 m$^2$/kg LW. Arable land occupation from the national herd was 16.1 ± 3.6 m$^2$/kg LW. Differences between supply chains occurred in response to relative feed conversion ratio (FCR) and differences in grain yield from region to region. The national average value was high compared with the CSF, primarily because the feed conversion was poorer than most of the CSF and because of the influence of production in South Australia, where yields tend to be lower than Victoria, NSW and Qld where many of the CSF were located.
Farm-gate scenario analyses

Table 6 shows the total fresh water consumption and fossil fuel energy demand for the Qld LC (baseline), and the alternative housing and manure management scenarios. Fossil fuel energy demand was reduced by 8%, 25% and 16% for the Qld LC – CAP with CHP (S1), Qld LC – CAP with CHP (S2) and Qld LC – DL (S3) respectively. Lower fossil fuel energy demand for the CAP with CHP scenario was the result of reduced electricity demand, and reduced gas use for heating at the piggery. Fresh water consumption remained the same as the baseline for the Qld LC – CAP with CHP (S1), Qld LC – CAP with CHP (S2) and Qld LC – DL (S3) scenario because of the lower requirement for cleaning water.

Wholesale pork – national herd

Fresh water consumption from the national herd was 184 ± 43 L/kg wholesale pork and was dominated by irrigation water for feed (74%), followed by the farm (20%) and meat processing (6%). Stress-weighted water use was 172 L ± 53 H₂O-e/kg wholesale pork. Fossil fuel energy demand from the national herd was 27 ± 2.6 MJ/kg wholesale pork. Energy demand was dominated by feed production (46.8%), followed by piggery energy use (23%), feed milling (16.2%) and meat processing (14%). Arable land occupation from the national herd was 25.9 ± 5.5 m²/kg wholesale pork.

Discussion

This study presents the first comprehensive analysis of resource use associated with Australian pork production using LCA, and one of few pork LCAs covering these impact categories internationally. Between CSF, impact intensity varied in response to housing system, feed efficiency and regional characteristics such as crop yields and the prevalence of irrigation water use in rations. Results were sensitive to a range of assumptions regarding feed grain production and diets and these are explored in the following section.

Sensitivity to model assumptions

Across most case studies and most impact categories, feed use was the greatest source of impacts, which was similar to studies investigating other environmental impacts in pig production systems (McAuliffe et al. 2016). As a consequence, the model was sensitive to several feed-related assumptions. Few data were available to compare energy, water or land occupation associated with different feed commodities. Comparing GHG emissions as a proxy for energy, we found few substantial differences between major grain inventory processes (derived from the AustLCI – Life Cycle Strategies 2015) and other Australian studies (Wiedemann et al. 2010; Brock et al. 2012). This suggests a reasonable level of agreement between grain LCI data for major Australian processes, where most energy sources are dependent on fossil fuels. To explore the impact of the inclusion rate of different cereal grains within rations, we
compared wheat and sorghum-based diets for Queensland, and wheat or barley diets in southern/western regions. We found no substantial effect on energy, water or arable land occupation from changing grain types between the major cereal grains. However, the inclusion rate of soybean meal did influence energy use, fresh water consumption and stress-weighted water use because imported soymeal is energy and water intensive. With all other factors remaining equal, diets utilising lower proportions of soymeal were found to have lower impacts.

Fresh water consumption and stress-weighted water use were sensitive to assumptions regarding irrigation rate and irrigation region. Inter-annual variation in irrigation rates in Australia can be high and thus water attributed to grain processes may differ from year to year. We found water use associated with grains to vary from 1.8 to 2.8 ML/ha between a low water availability year (2010) and a high water availability year (2008). We found that fresh water consumption varied by 22% below and 5% above the national herd value between high and low water use years. Inter-year variation in stress-weighted water use is expected to be very high in response to the different rates of irrigation used and the variable rates of extraction (and therefore water stress) from year to year, and estimates would be improved if annual, catchment specific WSI values were available. Arable land occupation was sensitive to grain yields. Inter annual variation in crop yields in Australia can be high and thus arable land attributed to grain processes differ from year to year. For example, national wheat yields varied from 2.0 (2006) to 0.9 t/ha (2007) (ABS2013) with processes differ from year to year. For example, national wheat speci

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high and thus arable land attributed to grain processes differ
from year to year. We found water use associated with grains
to vary from 1.8 to 2.8 ML/ha between a low water availability year
(2010) and a high water availability year (2008). We found that
fresh water consumption varied by 22% below and 5% above
the national herd value between high and low water use years.
Inter-year variation in stress-weighted water use is expected to be
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year to year, and estimates would be improved if annual, catchment
specific WSI values were available. Arable land occupation was
sensitive to grain yields. Inter annual variation in crop yields in
Australia can be high and thus arable land attributed to grain
processes differ from year to year. For example, national wheat
yields varied from 2.0 (2006) to 0.9 t/ha (2007) (ABS2013) with
the variation being a response to rainfall in each given year.
Taking this variability into account, arable land occupation
ranged from 27% below to 33% above the national herd value
when grain produced in these years was modelled. Considering
the sensitivity of water and land occupation results to a
specific year, benchmarking results require frequent updating
to remain current. Alternatively, averaging results over a longer
time period (3–5 years) is expected to produce more stable results.
This inter annual variation is a particular feature of Australian
systems, that are heavily influenced by climate variability. Further
research is required to understand the impact of this variability on
resource demand for grain crops and grain users such as pork.

Resource use and impact intensity
Fossil fuel energy demand ranged between 12.9 and 17.4 MJ/kg
LW at the farm-gate, and these results were similar to several
European studies (Basset-Mens and van der Werf 2005; Dourmad et al. 2014; Mackenzie et al. 2016). Energy demand
differed in response to energy efficiency at the piggery, housing
type and the energy intensity of the diet. Energy efficiency
research has identified substantial differences between farms
with different cooling systems in particular (McGahan et al. 2015),
suggesting that on-farm efficiency improvements are
possible, resulting in both environmental improvements and
cost savings. Interestingly, energy demand for pigs raised in
the WA O-DL system, were not lower than conventional
conventional production (WA LC). Although on-farm energy
demand was slightly lower for the O-DL system, this did not
compensate for the higher feed-related impacts associated with
poorer production efficiency (pigs weaned per sow per year and
FCR), which was similar to the finding of Dourmad et al. (2014).

In the eastern states, irrigation water embedded in feed
dominated fresh water consumption, rather than drinking water
at the piggery. Irrigation water varied widely between regions,
and was up to 86% lower in the WA supply chains because of
the small amount of irrigation used for grain production in this
state. Irrigation use and volume in grain production are not factors
that can be influenced by pig producers, though water use
associated with grain may be reduced by improving herd FCR
and this is one opportunity available to the industry to reduce
water use. Herd FCR is a major driver of production costs in
pig production, and is the subject of considerable focus from both
producers and researchers, via improvement in reproductive
efficiency, growth rates, nutrition and animal health. These
improvements are expected to yield both production benefits
and benefits regarding resource use efficiency and the
environment, provided trade-offs do not occur.

No LCA studies were found reporting fresh water
consumption using comparable methods to the present study,
though several water footprint studies have been completed
(Mekonnen and Hoekstra 2012; Gerbens-Leenes et al. 2013; de Miguel et al. 2015). These studies report ‘green’, ‘blue’ and
‘grey’ water, of which ‘blue’ water is broadly comparable to
fresh water consumption in the present study, though calculation
methods are quite different. Fresh water consumption in this
study (22.2–156.7 L/kg LW) was considerably lower than the
257–405 L/kg (converted to LW basis) reported by Mekonnen
and Hoekstra (2012) and de Miguel et al. (2015). These studies
found that intensive systems used less water per kg LW than
extensive systems due to the increased productivity of these
systems. An opportunity may exist to improve water-use
efficiency by improving the beneficial utilisation of effluent
at piggeries for crop production. Where this water can be used
to replace clean irrigation water, the apparent fresh water
consumption of the piggery could be reduced by up to 18%
by allocating this water to the end-user (crop) rather than the
piggery. Provided irrigation is managed within sustainable
limits, the added nutrients contained in pig effluent can also
reduce reliance on synthetic fertilisers in cropping systems
without substantial environmental burdens, though further
research is required in this area to understand eutrophication
risks associated with pig production.

Between CSF, regional water stress was found to have more
influence on the impact of water use than the total volume used;
suggesting the location of water use is a more important factor
governing impact on the environment than volume used. The
highest values were seen for the NSW and Vic. CSF, were stress-
weighted water use was higher than volumetric water use in
response to the high levels of water stress in the lower Murray–
Darling basin. Although this was a reasonable representation,
we note that the WSI values of Pfister et al. (2009) are of coarse
resolution, and may underestimate water stress in other regions.
Seasonal water stress should also be taken into account to
understand impacts more thoroughly. Further research is required
improve this analysis and apply updated methods developed
in this area.

No studies were found reporting stress-weighted water use for
pork production, but compared with beef production in eastern
Australia where stress-weighted water use ranged from 2.0
L–361.7 L H2O-e/kg LW (Wiedemann et al. 2016b) from
CSF, the stress-weighted water use from piggeries covered a smaller range of values.

Land occupation was dominated by feed production, whereas the land occupied for the piggery itself was insignificant because of the very high density of livestock on relatively small land areas. Arable land occupation for CSF ranged between 10.9 and 16.1 m²/kg LW, which was generally higher than values found in the literature, which ranged from 4.1 to 12.1 m²/kg LW (Basset-Mens and van der Werf 2005; Dournad et al. 2014; Bava et al. 2015; González-García et al. 2015). These comparison studies were predominantly from northern European countries, where grain yields are significantly higher than Australia, explaining the difference between these results and the present study.

Improvement scenarios
Energy generation from manure has been identified as an opportunity to improve efficiency in European systems (Nguyen et al. 2010). Scenario modelling in the present study revealed moderate reductions in energy demand for conventional housing systems, where the CAP and CHP were installed or deep litter housing systems used. One scenario (scenario 1) was based on an actual installed system at the case study piggery and therefore provides insight into the likely energy demand reduction under commercial conditions. The CAP with CHP system was installed at the grower-finisher site, where the largest amount of manure is produced. At this site, the heat produced by the CHP was not fully utilised, because it was logistically difficult to transport heat from the grower-finisher site to the breeder site where most heat is required. This is a common problem for multi-site piggeries and may limit the capability of CHP units to meet the heat requirements of the piggery. We also observed that transporting electricity from the generation site to other sites was logistically difficult; in some instances requiring the installation of privately owned power networks with high capital costs. Excess electricity was also not easy to sell by exporting to the grid at the Qld LC site, and at the time of performing the analysis, cost effective agreements had not been established with local power providers. As a result, excess biogas was flared to destroy the methane, but the energy potential was not utilised. Where we modelled all manure being treated in the CAP with CHP system (scenario 2), total energy demand per kg LW was reduced by 25%, to 13.1 MJ/kg LW.

We modelled deep litter housing for the grow-out and finishing stages. This scenario also provided substantial reductions in energy demand (16%) and fresh water consumption (28%). The reduction in energy demand is primarily a function of reduced electricity consumption using the deep litter housing system, as it was not tunnel ventilated as the conventional system was. The decrease in fresh water consumption was primarily influenced by the reduction of cleaning and cooling water at the piggery. According to the National Greenhouse Gas Inventory, 6.5% of manure was treated in CAP or engineered digesters in the 2010–2013 period (derived from Commonwealth of Australia 2015). A large proportion of manure is currently treated in anaerobic effluent systems (71.4%, see Wiedemann et al. 2016a; Supplementary material).

The introduction of government payments in Australia to reduce greenhouse gas emissions has made installation of anaerobic digestion and energy production equipment much more cost effective, as evidenced by the substantial number of Australian piggeries installing this equipment.

Incremental improvements in environmental performance may also be achieved by improving feed efficiency in the long-term. Whole herd FCR (LW) ranged from 2.4 to 3.2 in the present study with an average of 2.9. This corresponds well to the national overall average of 2.96 reported by the industry in 2010–2011 (Pork CRC, unpubl. data). Feed efficiency has improved in Australia, with a 5% reduction in FCR being recorded between 2010 and 2015 (Pork CRC, unpubl. data). Provided diets and other inputs remain similar, this improvement should contribute to lower energy, water and arable land requirements for pork production, reducing impacts on primary resources. As feed is also a major production cost, improvements in FCR provide economic and environmental benefits. Though not studied here, reduced crude protein diets have been shown to result in lower environmental impacts in pig systems elsewhere in the world (McAuliffe et al. 2016) and although this approach tends to aim at reducing greenhouse gases (i.e. Wiedemann et al. 2016c) and nutrient-related impacts, it may also offer opportunities to reduce resources if FCR is also improved or if the reliance on high impact commodities such as soymeal is reduced. Further analysis of this would be beneficial to industry. Considering the significance of feed impacts, improvements may also be achieved by utilising higher proportions of by-product feeds in pig rations. Pigs have an important role in utilising low value by-products or waste products from the human food supply chain and further investigation into the environmental significance of including these by-products is warranted.

Wholesale pork – national herd
When impacts were assessed through to the point of a wholesale ready product, impacts were dominated by feed production (63–99% of total impacts) and on-farm production (0.5–23%), whereas meat processing tended to be a smaller contribution (0.5–14%). Because of the mass losses associated with meat processing, reported impacts per kilogram of pork rose substantially compared with LW. Comparison pork results that included meat processing were more difficult to find. Mackenzie et al. (2016) and Mekonnen and Hoekstra (2012) provide results on an expected carcass weight basis but they do not include impacts from meat processing. González-García et al. (2015) included the role of meat processing and, similar to the present study, found the contribution to generally be <5%. These authors reported fossil energy demand of 30.9 MJ/kg edible product, whereas arable land occupation was 8.5 m²/kg edible product, calculated with an edible yield of 62% of carcass weight. The wholesale product used as the FU in our study was a ‘bone-in’ product, with an estimated edible yield of 85%. Relative to carcass weight, this resulted in 69% edible yield in the present study. This slightly higher value may be the result of skin and subcutaneous fat, which were included in the edible portion on some cuts. When impacts were recalculated and reported per kilogram of edible meat (bone-free), they were 32 MJ energy,
212 L fresh water, 203 L H₂O-e and 31 m² land/kg pork, which was similar to the energy values reported by González-García et al. (2015) whereas land was much higher in the present study. Regional averages for beef and lamb produced in eastern Australia showed lower energy demand for lamb and similar energy demand for grass fed beef, whereas stress-weighted water use (108.5–169.4 L H₂O-e/kg edible meat) and arable land use was lower for lamb and grass-fed beef (Wiedemann et al. 2015c) compared with the present analysis of Australian pork. The lower stress-weighted water use associated with lamb and beef is a function of production in lower water-stressed regions, compared with the high impacts associated with irrigation water use in the pork supply chain. The lower requirement for arable land in lamb and beef production reflects the predominance of rangeland grazing in these industries, which relies on non-arable land. All impacts trended higher than chicken meat production in two Australian regions (Wiedemann et al. 2017), largely in response to the higher feed requirements for pork compared with chicken meat.

Conclusions

This study provides the first case study and national analysis of energy, water and land occupation from Australian pork production using LCA. Australian pork tended to utilise similar amounts of fossil fuel energy and smaller volumes of fresh water than northern hemisphere production systems. However, Australian production typically had larger requirements for arable land because of the lower yields in Australian cropping regions. The impact on water use was found to be lower than the volumetric volume may suggest in several regions, but not for the national average production. However, noting the coarse scale of the WSI values applied, further research is required to apply new water stress impact methods. The study found that feed production is the dominant contributor through to production of a wholesale product for energy, water and land occupation. Considering the importance of feed impacts for all impact categories, FCR is an important production metric influencing the environmental efficiency and cost of production for pork and improvements in this metric are expected to reduce resource-related impacts. Alternative manure management systems that produce energy from biogas and in some cases, alternative housing systems, may also be used to reduce energy and water from pork production. Considering that changes may occur in system efficiency, FCR, manure management and grain production over time, and considering the impact of seasonal variations, it is recommended that benchmark results are produced at 5-yearly intervals to maintain the currency of the results.

Acknowledgements

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FAO (2009) ‘How to feed the world in 2050.’ (Food and Agriculture Organization of the United Nations: Rome)

Gerbens-Leenes P, Mekonnen M, Hoekstra A (Food and Agriculture Organization of the United Nations: Rome)


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### 9.3 ADDENDUM

The system boundaries applied in this study (both papers) were consistent with the summary tables presented in the discussion (Table 4 and Table 5). However, for the case study results that were previously only presented at the farm gate, the system boundary was extended to report results relative to boneless meat. This was achieved using assumptions applied to the national supply chain, as reported in the paper.

Human edible protein required (HEPR) was not reported in this study and was calculated to produce Figure 3, section 10.5.7.

The HEPR was determined by calculating the weighted average percentage of human edible protein in each diet, relative to one kilogram of boneless meat produced. Weighted average human edible diet protein was 13.8%, and total kilograms of grain per kilogram of boneless meat was 5.9 kg, resulting in human edible protein of 0.81 kg per kilogram of boneless pork. The protein percentage of boneless pork was assumed to be 19% and HEPR was 4.27.
10 DISCUSSION

10.1 RESOURCE AND IMPACT HOTSPOTS IN MEAT PRODUCTION SYSTEMS

The published studies presented in this thesis cover > 80 CSF and 19 processing facilities across all species. Regional analyses were performed for pork, chicken meat, beef and lamb, covering major Australian production regions and accounting for > 35% of national production of the respective industries, allowing regional influences to be assessed for each industry respectively. These regional analyses for the grazing industries typically averaged data over five years to minimise the impact of climatic variability (aligning with the recommendations of Harris and Narayanaswamy, 2009a) and limitations resulting from averaging these data over an extended timeframe are discussed. Results are shown in Table 4 and Table 5, with common system boundaries and reference flows reflecting ‘wholesale’ meat products. This enabled a benchmarking and hotspot analysis to be completed, which informed an investigation of mitigations and trade-offs based on the available case studies. It is noted that this dataset did not enable comparison at a national level because a nation-wide analysis of chicken meat and lamb was not completed, but regional influences were analysed and the implications for the whole industry are discussed. Within industries, the major different production systems were also assessed. These can be variously grouped into ‘conventional or alternative’ production systems, or by breed (i.e. Merino vs meat breed) or market type (i.e. FR vs conventional meat chickens). However, to examine the impact of trade-offs in the present study, the production systems have been grouped according to production intensity instead. Here production intensity relates to the level of inputs to the system, which may include management or labour, purchased inputs or capital inputs such as built infrastructure, or stocking rates in grazing systems. Many of these aspects operate along a continuum, and this is noted in the discussion without strictly defining the level of intensity. It must be noted that broad trends were drawn from the available results, which in most cases did not include paired trials. Consequently, the synthesis is limited to a discussion of the observable trends. In some cases, insufficient data were available to inform the synthesis, and these were noted as limitations.
Table 4. Environmental impacts and resource use for Australian chicken meat and pork

NOTE: Results adjusted to “per kg of boneless meat” including impacts from meat processing using data from the author’s studies

<table>
<thead>
<tr>
<th>Study</th>
<th>Region or state</th>
<th>Production system</th>
<th>Fossil energy (MJ)</th>
<th>Water use (L)</th>
<th>Water stress (L. H2O-e)</th>
<th>Crop land (m²)</th>
<th>Total land (m²)</th>
<th>GHG (kg CO2-e)</th>
<th>GHG (LU, dLUC) (kg CO2-e)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chicken meat</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wiedemann et al. (2017b)</td>
<td>QLD</td>
<td>conventional</td>
<td>29.8</td>
<td>154.7</td>
<td>36.4</td>
<td>19.5</td>
<td>19.5</td>
<td>3.1</td>
<td>1.7</td>
</tr>
<tr>
<td></td>
<td>SA</td>
<td>conventional</td>
<td>25.1</td>
<td>52.7</td>
<td>50.7</td>
<td>31.3</td>
<td>31.3</td>
<td>2.5</td>
<td>1.4</td>
</tr>
<tr>
<td></td>
<td>QLD/SA</td>
<td>free range</td>
<td>25.5</td>
<td>96.8</td>
<td>29</td>
<td>25.3</td>
<td>25.3</td>
<td>2.5</td>
<td>0.6</td>
</tr>
<tr>
<td>Pork</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wiedemann et al. (2016b; 2018b)</td>
<td>QLD</td>
<td>Small-medium conventional</td>
<td>37.3</td>
<td>254.0</td>
<td>28.8</td>
<td>23.4</td>
<td>23.4</td>
<td>9.3</td>
<td>1.1</td>
</tr>
<tr>
<td></td>
<td>QLD</td>
<td>Large conventional</td>
<td>38.2</td>
<td>315.1</td>
<td>28.0</td>
<td>20.6</td>
<td>20.6</td>
<td>8.2</td>
<td>1.3</td>
</tr>
<tr>
<td></td>
<td>QLD</td>
<td>Conventional-deep litter</td>
<td>33.3</td>
<td>213.5</td>
<td>285.6</td>
<td>28.9</td>
<td>28.9</td>
<td>4.4</td>
<td>0.4</td>
</tr>
<tr>
<td></td>
<td>VIC</td>
<td>Large conventional</td>
<td>32.2</td>
<td>261.2</td>
<td>329.9</td>
<td>24.4</td>
<td>24.4</td>
<td>7.2</td>
<td>1.3</td>
</tr>
<tr>
<td></td>
<td>WA</td>
<td>Large conventional</td>
<td>28.3</td>
<td>44.6</td>
<td>13.0</td>
<td>23.8</td>
<td>23.8</td>
<td>7.6</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>WA</td>
<td>Outdoor-deep litter</td>
<td>30.0</td>
<td>51.5</td>
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<td>30.4</td>
<td>30.4</td>
<td>4.7</td>
<td>0.2</td>
</tr>
<tr>
<td>National</td>
<td></td>
<td>average of all systems</td>
<td>31.8</td>
<td>216.2</td>
<td>202.4</td>
<td>30.5</td>
<td>30.5</td>
<td>6.5</td>
<td>0.9</td>
</tr>
</tbody>
</table>
## Table 5. Environmental impacts and resource use for Australian beef and lamb products

**NOTE:** Results adjusted to “per kilogram of boneless meat” including impacts from meat processing using data from the author’s studies

<table>
<thead>
<tr>
<th>Study</th>
<th>Region or state</th>
<th>Production system</th>
<th>Fossil energy (MJ)</th>
<th>Water use (L)</th>
<th>Water stress (L H₂O-e)</th>
<th>Crop land (m²)</th>
<th>Total land (m²)</th>
<th>GHG (kg CO₂-e)</th>
<th>GHG (LU, dLUC) (kg CO₂-e)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Lamb</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Wiedemann et al.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(2016f)</td>
<td>VIC CSF*</td>
<td>specialist export lamb</td>
<td>22.0</td>
<td>162.6</td>
<td>16.4</td>
<td>3.2</td>
<td>64.4</td>
<td>17.5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>VIC RAF</td>
<td>specialist export lamb</td>
<td>24.4</td>
<td>593.4</td>
<td>232.4</td>
<td>6.4</td>
<td>133.5</td>
<td>19.6</td>
<td>-1.0, 0.3</td>
</tr>
<tr>
<td></td>
<td>NSW CSF</td>
<td>specialist export lamb</td>
<td>16.7</td>
<td>163.9</td>
<td>55.8</td>
<td>3.2</td>
<td>95.9</td>
<td>19.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>NSW RAF</td>
<td>specialist export lamb</td>
<td>21.9</td>
<td>533.3</td>
<td>368.1</td>
<td>2.4</td>
<td>562.1</td>
<td>20.4</td>
<td>-0.9, 0.2</td>
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<td></td>
<td>SA CSF</td>
<td>specialist export lamb</td>
<td>14.1</td>
<td>269.8</td>
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<td>0.5</td>
<td>186.5</td>
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<td></td>
<td>SA RAF</td>
<td>specialist export lamb</td>
<td>26.3</td>
<td>644.8</td>
<td>24.1</td>
<td>17.1</td>
<td>361.3</td>
<td>18.3</td>
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<tr>
<td>Wiedemann et al.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(2016e)</td>
<td>HRZ CSF*</td>
<td>Merino / Merino cross</td>
<td>18.3</td>
<td>211.4</td>
<td>74.6</td>
<td>3.0</td>
<td>115.1</td>
<td>22.3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>HRZ RAF</td>
<td>Merino / Merino cross</td>
<td>23.7</td>
<td>189.5</td>
<td>67.5</td>
<td>6.1</td>
<td>8.8</td>
<td>20.8</td>
<td>-1.4, -0.5</td>
</tr>
<tr>
<td></td>
<td>WSZ CSF*</td>
<td>Merino / Merino cross</td>
<td>22.9</td>
<td>315.3</td>
<td>12.1</td>
<td>34.5</td>
<td>105.2</td>
<td>19.4</td>
<td></td>
</tr>
<tr>
<td></td>
<td>WSZ RAF</td>
<td>Merino / Merino cross</td>
<td>28.3</td>
<td>342.5</td>
<td>13.1</td>
<td>47.1</td>
<td>256.7</td>
<td>18.9</td>
<td>-0.1, 0.0</td>
</tr>
<tr>
<td></td>
<td>SPZ CSF*</td>
<td>Merino / Merino cross</td>
<td>15.1</td>
<td>295.4</td>
<td>8.9</td>
<td>0.0</td>
<td>2309.8</td>
<td>18.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>SPZ RAF</td>
<td>Merino / Merino cross</td>
<td>19.3</td>
<td>347.4</td>
<td>10.5</td>
<td>0.9</td>
<td>8005.6</td>
<td>18.8</td>
<td>-1.7</td>
</tr>
<tr>
<td>Wiedemann et al.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(2016a)</td>
<td>QLD CSF</td>
<td>Export (n=6)</td>
<td>19.8</td>
<td>640.0</td>
<td>153.7</td>
<td>0.7</td>
<td>946.6</td>
<td>27.3</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>Domestic (n=3)</td>
<td></td>
<td>20.9</td>
<td>593.9</td>
<td>19.5</td>
<td>3.1</td>
<td>507.4</td>
<td>26.9</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>QLD RAF</td>
<td>Export</td>
<td>22.1</td>
<td>754.3</td>
<td>227.3</td>
<td>3.8</td>
<td>806.5</td>
<td>28.0</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>NSW CSF</td>
<td>Export (n=5)</td>
<td>24.5</td>
<td>460.6</td>
<td>101.6</td>
<td>14.3</td>
<td>120.4</td>
<td>24.9</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>Domestic (n=5)</td>
<td></td>
<td>20.9</td>
<td>274.4</td>
<td>80.6</td>
<td>6.0</td>
<td>120.4</td>
<td>24.1</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>NSW RAF</td>
<td>Export</td>
<td>26.1</td>
<td>474.4</td>
<td>37.0</td>
<td>3.4</td>
<td>287.5</td>
<td>27.8</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td>Domestic</td>
<td></td>
<td>25.4</td>
<td>449.1</td>
<td>35.6</td>
<td>3.1</td>
<td>276.1</td>
<td>26.7</td>
<td>n.r</td>
</tr>
<tr>
<td>Wiedemann et al.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(2015c)</td>
<td>National average</td>
<td>National average</td>
<td>33.5</td>
<td>1162.8</td>
<td>n.r</td>
<td>n.r</td>
<td>n.r</td>
<td>29.5</td>
<td>n.r</td>
</tr>
<tr>
<td>Wiedemann et al.</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>(2017a)</td>
<td>Grain finished</td>
<td>SF* (Domestic)</td>
<td>29.7</td>
<td>672.8</td>
<td>222.2</td>
<td>12.5</td>
<td>238.1</td>
<td>22.6</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td></td>
<td>MF* (export)</td>
<td>33.2</td>
<td>699.7</td>
<td>264.7</td>
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<td>218.0</td>
<td>21.5</td>
<td>n.r</td>
</tr>
<tr>
<td></td>
<td></td>
<td>LF* (export)</td>
<td>47.8</td>
<td>471.5</td>
<td>41.0</td>
<td>53.7</td>
<td>226.9</td>
<td>24.2</td>
<td>n.r</td>
</tr>
</tbody>
</table>

*CSF: Case study farm; RAF: Regional average farm; HRZ: High rainfall zone of NSW; WSZ: Wheat sheep zone of WA; SPZ: Southern pastoral zone of SA.

n.r, not reported in the original study but was investigated by the author and colleagues in a separate publication (Henry et al., 2015c). See discussion in the following sections. *SF: short-fed, *MF: mid-fed, *LF: long-fed.
**10.2 REGIONAL INFLUENCES**

The published studies typically selected major production regions as a focus for the analysis, but the scope of most studies did not extend to a nation-wide assessment. Specifically, no national analyses exist for lamb or chicken meat, but a national assessment was done for pork (chapter 9), and the author has elsewhere published national results for the beef slaughter herd in a trends analysis from 1981-2010 (Wiedemann et al., 2015a - see supplementary material).

Region and intensity of production (discussed in section 10.2) are correlated in some instances in Australia, for a range of reasons. While the published studies did not test the impact of each factor independently, a broad assessment can be made by examining the regional examples and the trends between different meat species. Causal relationships between impacts and region such as climate driven effects were assessed.

### 10.2.1 PORK AND CHICKEN MEAT

Energy demand was found to differ between supply chains in response to production system type (see section 10.3) and to some extent, regional effects. Queensland production systems utilised the highest amount of energy of all the regions studied, largely because tunnel ventilation is more common (McGahan et al., 2014b) in the region to manage higher temperatures. Deep litter systems were less prevalent (Commonwealth of Australia, 2016) because of climate related limitations to pig productivity with these systems. Energy demand for chicken meat production was higher in QLD compared to SA because diets in this state utilised higher levels of imported soymeal.

Regional differences in water use were also apparent, with the WA pork production and SA chicken meat production having substantially lower water use because less irrigation was used in the feed supply system. Interestingly, this did not necessarily translate to lower water impacts. In SA, chicken meat water stress was higher than in QLD, because this is a highly water stressed region. Similarly, while water use in the
pork systems varied by a factor of 4-6 between irrigation and non-irrigation regions, differences in water stress were even more pronounced and were only partly explained by the volume of water used. For example, water stress was lowest in WA and was highest in QLD, but the differences were much less than the volume of water may suggest. This was because of the relatively low WSI in these regions of QLD compared to the national average and NSW and VIC supply chains. Water stress was very high for the NSW and VIC supply chains because volumes and WSI were both high. This largely explained the high water stress for the national average, because approximately 46% of the Australian pig herd is located in south eastern Australia, where irrigation in grain production and water stress is high. This finding also suggests that water use and water stress may be higher for ‘average’ chicken meat in Australia than shown by the case studies in chapter 8. This is because a large proportion of chicken meat production is located in NSW and VIC, where utilisation of irrigation for grain in high water stress regions is prevalent. The study in chapter 8 investigated this by modelling a typical NSW diet, which showed much higher levels of water stress than either of the main case studies. This finding suggests that further research is therefore needed to accurately understand water use and water impacts in Australian chicken meat production.

Trends in crop land were less clear between regions in the pork study, because this was masked by differences in herd productivity (feed conversion ratio - FCR). However, there were also regional differences in cereal crop yield. For example, wheat yields are historically higher in Queensland than in SA (ABS, 2013) and this contributes directly to the difference in land occupation for chicken meat in these states. This influence can be seen more clearly with chicken meat production where crop land for QLD production was 38% lower than SA, despite FCR being similar.

Greenhouse gas emissions from pork production were influenced by region via the slightly higher MCFs applied for QLD compared with the southern states. As noted previously, LU and dLUC emissions were also higher for QLD chicken meat, and this was observed in QLD pork production, though some south eastern supply chains had similarly high inclusion rates of soymeal and concomitant LU and dLUC impacts. A clear difference was observed in both chicken meat (SA) and pork (WA)
where LU and dLUC from imported soymeal was minimal, leading to lower impacts from these regions.

10.2.2 BEEF AND LAMB

With respect to grazing systems, substantial regional differences in energy demand were observed, in response to differences in system intensity. In contrast, regional impacts were observed in water use and water impact, with the availability of irrigation water, climatic effects (temperature and net evaporation rates) and water supply sources all influencing the results. Irrigation water use is regionally specific and may have a disproportionately large influence on results. For example, a moderate pasture irrigation rate (3 ML ha\(^{-1}\) yr\(^{-1}\)) producing a large amount of beef (for example, 800 kg LW ha\(^{-1}\) yr\(^{-1}\)) will utilise 3750 L kg LW\(^{-1}\). No case studies in Australia were included that used irrigation rates this high. However, the national study of Wiedemann et al. (2015a) found that irrigation contributed 30% of the national average water use for the beef slaughter herd and contributed 152L kg LW\(^{-1}\) to every kilogram of beef produced for slaughter in Australia. This suggests that some regions utilise large volumes of irrigation water for beef cattle production, though further research with regional scale water inventories is needed to understand the impact of this water use at the regional level. In addition to spatial variability, water use can also be temporarily variable (see chapter 8, and Peters et al. 2010b). This highlighted a potential weakness in averaging regional data over a long time period, because inter-year influences were not considered in detail but may have a significant effect on production systems from year to year.

South eastern lamb production regions were also observed to have higher water use associated with irrigation (RAF datasets) when averaged over several years. High temperatures and high net evaporation rates were found to influence water use in regions that relied on farm dams for livestock water supply. For example, the WA wool production region (chapter 7) had higher net evaporation rates than the NSW high rainfall region in the same study. Despite the latter having much more on-farm water storage relative to livestock numbers, evaporation rates were higher in WA and this led to much higher water use demonstrating the regional effect. Interestingly,
there were factors that mitigated the high net evaporation rates in some regions. Comparing the same results to the SA pastoral zone (wool, chapter 7) showed this region had very high net evaporation rates and warm temperatures in summer, but a large proportion of water was supplied from bores. Water use in SA was consequently similar to the WA region of the same study (where dams predominated), despite the lower evaporation rates in the latter. This shows regions can adapt to manage water resource constraints, though this depends on the availability of an alternative such as ground water.

Strong regional influences were observed with respect to crop land for beef and lamb production. In beef production, the NSW CSF (but not RAF) had much higher crop land occupation than the QLD systems. This trend was also strongly demonstrated in comparison of the NSW and WA wool systems, and in a comparison of the VIC CSF and the SA CSF lamb systems. Clearly, availability of arable land varies between regions and is highest in mixed cropping and livestock areas such as the WA region studies. This finding indicates that results for beef and wool are limited because major mixed cropping regions in south western NSW and VIC have not been assessed. This is a remaining knowledge gap in the research.

Regional influences on GHG emissions were apparent in some regions, where the combination of rainfall and soil type enabled better herd or flock production. However, this effect was not always easy to separate from production intensity (section 10.3), because high rainfall regions with fertile soils often receive higher inputs, becoming ‘intensive’ grazing regions. It is likely that the regional differences would be more pronounced if the contrast between grazing regions was greater, and this has been shown by other supporting research. Wiedemann et al. (2015a) found emissions intensity was higher in some northern beef regions, leading to a higher estimated emission intensity than observed in most of the case studies examined here. For example, Wiedemann et al. (2015d) found emissions intensity was 16.2-17.5 kg CO$_2$-e kg LW$^{-1}$, exclusive of energy related emissions, in some Northern Territory herds. This was 60% higher than the lowest CSFs, located in high rainfall southern regions (chapter 3).
Regional influences were even more substantial for LU and dLUC in grazing beef. Emissions from land clearing are highly variable between states in Australia, predominantly because of policy differences between states and development history. Emissions associated with LU and dLUC were found to be high for some beef production regions in QLD, where LU and dLUC emissions potentially exceeded the combined emissions from all other sources (Henry et al., 2015c). But these results were highly sensitive to the assumptions used for the carbon stocks lost, regrowth sequestration and the timeframe covered. While historic emissions (1990-2010) were high (ranging from 0.246-0.445 t CO2-e ha⁻¹ year⁻¹), emissions post 2006 were estimated to be lower and potentially negative, indicating sequestration, ranging from -0.074 to 0.11 t CO2-e ha⁻¹ year⁻¹ (Henry et al. 2015b). However, future emissions are influenced by vegetation regulations that can change in response to policy decisions (Reside, 2017) and forecast estimates therefore have high uncertainty. The results for NSW regions in the same study provided a strong contrast, where LU and dLUC emissions were estimated to range from -0.12 to 0.033 t CO2-e ha⁻¹ year⁻¹ (1990-2010). This corresponded to emissions ranging from -1.4 to 0.3 kg CO2-e kg LW⁻¹ in this region. Because LUC is a political issue, regional variation is likely to continue in Australia for some time, though emissions associated with beef are clearly diminishing over time (Wiedemann et al., 2015a). The regional differences were also influenced by the methodological choice to amortise impacts over a 20 year period, and while data were not available to examine differences between states over a longer time period, it is likely that the differences would diminish if this analysis was done. Similarly to irrigation water use, dLUC impacts in relatively small regions can have a disproportionately large influence on regional or national results and as a consequence, regional representativeness is important for ensuring representative results. Future research is required to understand these emissions for the Australian beef industry.

10.2.3 SUMMARY OF KNOWLEDGE: REGIONAL EFFECTS

A series of reasonably consistent regional effects was observed between species and impact categories. Energy demand was generally not affected by region when differences in production system were discounted, except in the case of slightly
higher energy for ventilation in QLD pork production. Water use and water stress was found to be highly regionally sensitive, particularly with respect to irrigation, and also because of climatic impacts on water losses. Irrigation varied between regions and this had a large effect on water use and water stress for pork, chicken meat and lamb. This also revealed a knowledge gap with respect to irrigation in beef systems of southern Australia.

Water stress and crop land were the most sensitive factors with respect to region. Water stress is calculated using ‘stress weightings’ that multiply impacts by factors ranging between 0.01 and 1, generating very large differences between regions. For different reasons, crop land also varied substantially. In comparison of regional pork and chicken meat production this was a function of crop yield, as demonstrated by the relative crop land use in QLD and SA chicken meat. In lamb and beef, grazing in mixed farming regions utilised large amounts of arable land because of the higher relative proportion of this resource. These assessments also showed large areas of arable land used for pasture, which may have been part of pasture leys between cropping or may have represented land that was marginal for cropping because of soil infertility or other factors. Thus, these arable pasture land areas should be considered as interchangeable with crop land without taking into account the risk of trade-offs from changes in LU, GHG emissions and soil health.

Region had little effect on GHG emissions from intensive livestock systems because these were less influenced by differences in climate. Emissions from lamb and wool systems also showed little variation between regions, though this observation may have been constrained by the number of regions investigated and the degree of variation between them. In contrast, impacts were found to vary substantially for beef when other associated studies were considered. Additionally, LU and dLUC impacts were highly regionally sensitive for beef and were moderately sensitive for chicken meat and pork, but less so for lamb. Further research is warranted to understand and utilise differences in LU and dLUC, to mitigate impacts from livestock rather than being seen only as a source of emissions. Southern high rainfall regions may have the potential to mitigate small amounts of GHG emissions (see chapter 7) or even large amounts of GHG emissions via sequestration. For example, Doran-Browne et al. (2016) demonstrated that a grazing farm in south-eastern
Australia could sequester adequate carbon in soils and trees to be carbon neutral over a long period of time. To summarise the broad trends observed in this analysis, Table 6 below provides a graphical representation of findings from the published studies.

### Table 6. Comparison of the impact of production region on resource use (energy, water, crop land, HEPR) and environmental impacts (GHG, LU, dLUC, water stress)

Notes: Water use is defined as fresh water consumption. HEPR is human edible protein required. LU, dLUC Land use and direct land use change GHG emissions. White indicates no change in impacts. Grey indicates that region does have an effect on this impact.

<table>
<thead>
<tr>
<th>Species (meat type)</th>
<th>Energy</th>
<th>Water use</th>
<th>Water stress</th>
<th>Crop land</th>
<th>GHG</th>
<th>LU, dLUC</th>
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<td>Pork</td>
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Noting that only 8 complete CSF were found in the Australian LCA literature prior to the research in this thesis and most only covered one impact, severe limitations existed with respect to the representativeness of the results from these studies. This has been improved by the increased number of CSFs presented here, and further improved via the regional analyses that were completed to improve representativeness. However, considering the regional sensitivity of many impacts, the findings are not comprehensive. One important implication of this finding is that researchers and decision makers should be careful in applying findings from one region to another or applying mitigations from one region to another, without taking into account the potential for trade-offs in each region.

### 10.3 PRODUCTION INTENSITY

#### 10.3.1 PORK AND CHICKEN MEAT

Three major pork production systems exist in Australia: conventional housing with effluent ponds (with and without biogas, combined heat and power), deep litter sheds and outdoor production. The proportion of pigs in each system has changed over time. The author’s NIR review (Wiedemann et al., 2014c) identified an increase in
the number of outdoor pigs from approx. 2% to 5% over the 25 years to 2014, while
over the same period deep litter housing increased from approx. 1% to >20% in
2014, with most of the remaining pigs being housed in conventional sheds with
effluent flushing.

Conventional production is the benchmark system in Australia and is the most
intensive in terms of capital investment and inputs. Deep litter (housing growing pigs
on straw or sawdust) may be termed an intermediate system in terms of production
intensity. Capital costs are lower and some inputs are lower than conventional
production but pig performance is similar. Outdoor production is most commonly
only used for breeding in Australia and is less intensive than conventional production
in terms of capital and some other inputs but is quite labour intensive. Most pigs bred
in outdoor production systems are finished on deep litter in Australia and are
transferred at weaning (3-4 weeks).

Based on results in section 9.1 and 9.2, deep litter sheds in comparison with
conventional sheds were found to have lower GHG emissions (less 38%), energy
demand (less 16%) and water requirements (less 28%). GHG emissions were lower
because manure was handled in predominantly aerobic conditions in deep litter
sheds, compared to the anaerobic effluent treatment systems used in conventional
sheds, reducing emission potential from the volatile solid (VS) fraction of the
manure. Energy demand was reduced because there was less energy required for
flushing and cleaning, and water use was lower because water was not required for
flushing manure.

Diet and pig performance (specifically FCR) was similar for conventional and DL
production, and consequently up-stream impacts related to feed were not changed,
excluding regional effects. These results show the less intensive system can deliver
better outcomes in terms of environmental impact and resource use. However, deep
litter sheds are often less preferred by the industry because pigs are harder to handle
and therefore variability in weight and condition is sometimes observed in the
slaughter pigs (NSW DPI, 2006).
Outdoor and conventional production systems were studied in WA, where outdoor production is prevalent. This analysis revealed lower GHG emission impacts because manure was deposited directly to soil and was therefore handled in a predominantly aerobic environment compared to conventional anaerobic manure treatment. However, crop land occupation was 28% higher in outdoor production and energy demand was slightly (6%) higher, while water use and water stress were similar. These trends were driven by the poorer FCR for outdoor production and the higher requirements for land on-site. Because land occupation was higher, further investigation is required in regions where this may influence LU or dLUC emissions. Impacts from high nutrient loading rates in outdoor systems also require further consideration (Wiedemann, 2015) to avoid trade-offs.

Two production systems are prevalent for meat chickens in Australia; conventional housing, where birds are housed on the floor of the shed with bedding material, and free range (FR) where birds are allowed to range after the brooding phase. Free range production represents approximately 15% of production (ACMF, 2011) with a small fraction of this also being ‘organic’. Conventional production is the remaining proportion. The results in chapter 8 showed a trend of lower energy demand for FR production, and this was clearly evident when the results were disaggregated by state. Impacts associated with feed, including water, crop land, GHG emissions, LU and dLUC were confounded by aggregating the FR data in the study reported in chapter 8. However, when this impact was removed there was no substantial variation between the two systems, because FR production had similar FCRs. The similar FCRs for FR and conventional meat chickens contrasted with both Australian egg research by the author (Wiedemann and McGahan, 2011) and international research (Leinonen et al., 2012a; Leinonen et al., 2012b; Williams et al., 2006) which showed poorer FCRs for FR production. The comparatively good FCR of Australian meat chickens may be because birds are not allowed to range until after the brooding period and may therefore range less, resulting in better FCRs because less energy is exerted in exercise than in the European production systems. The results for resource use contrast with Australian egg industry research, which showed higher energy demand and GHG emissions for FR egg production largely because of the impact of poorer FCRs in FR production (Wiedemann and McGahan, 2011), and this was similar to international research by Williams et al. (2006). Similarly to outdoor pigs,
nutrient risks may exist for FR production, though risks were found to be low for Australian egg production (Wiedemann et al., 2018a) and may be lower in FR meat chickens because of the different ranging behaviour (Brown and Gallagher, 2015). Further analysis of this aspect is required in LCA to understand the impacts on an intensity basis and avoid trade-offs.

10.3.2 BEEF AND LAMB

Differences were also observed between the sheep and beef case studies with respect to the intensity of production, though, as noted, the studies were not paired or controlled, allowing a broad consideration of these effects only.

Comparison of lamb production between states revealed that grazing intensity was higher in Victoria CSF, in terms of stocking rates and purchased inputs, than the NSW CSF. Victorian production had higher weaning rates, slaughter weights and wool production, which corresponded to 12% lower GHG emissions compared to NSW. However, these differences were accompanied by higher energy inputs (+58%) associated with higher on-farm fuel use and fertiliser inputs, and higher utilisation of arable land for pasture and crop production (+222%), suggesting that the gains were achieved by utilising better quality land, and by increasing inputs to improve pasture quality and supply, with consequent benefits for livestock production. Water use was not strongly influenced by production intensity and did not differ substantially between the CSFs. However, the regional data showed higher irrigation water use in VIC compared to NSW and total water was slightly higher. This did not correspond to water stress, which was more regionally influenced (see section 10.2). In contrast to the trend for arable land, total land occupation was lower in VIC because of the higher stocking rates.

Comparison between the intensive VIC CSF with the extensive grazing system (SA pastoral CSF) provided an interesting contrast. Production intensity was dramatically higher in the Victorian CSF, and this corresponded to higher energy demand and crop land utilisation. In contrast, total land occupation was much higher in SA
because of the low stocking rates. Water use was also low in VIC compared to SA, but this corresponded more to regional (climatic) factors. While water stress was low for both regions, it was notable that this was higher in VIC despite the low volume of water used. Interestingly, GHG emissions were very similar in both systems, demonstrating that extensive grazing systems with low inputs can still achieve reasonable outcomes in terms of emissions intensity. This was partly because of the lower manure and CO₂ from energy in the SA systems.

Another comparison was provided by the wool study. The NSW production region had higher stocking rates than WA, but the WA production region was a mixed farming zone and had much higher inputs of arable land, direct farm energy and grain, making it a more intensive production system. This resulted in higher energy demand for the WA farm and lower GHG emissions (comparing CSFs) because flock productivity was better in WA than in NSW. Differences in sheep genetics were also noted in this region; the WA CSF produced more cross-bred lambs from Merino ewes, increasing meat production per breeding ewe, while the NSW CSF produced finer wool. These trends in sheep genetics were not evident in the RAF. Water use was higher in the WA region, but this related more to regional factors (see section 10.2). These results show higher resource use and corresponding lower GHG emissions in more intensive production regions.

Comparison of beef production systems revealed mixed results. More intensive systems in southern Australia had higher energy demand in most cases, and higher arable land requirements (CSF), though regional average crop land was not different between the regions. The more intensive farms tended to have lower GHG emissions (CSFs) because of better herd performance, but this was not supported by the regional analysis where inputs of energy and land were high, but herd performance was not sufficient to reduce livestock GHG emissions. There were also examples of extensive farms, such as those located in higher rainfall parts of QLD, which had very large land areas, relatively low inputs and very good herd performance, resulting in low GHG emissions. This was similar in some respects to the SA extensive lamb production systems and demonstrated that extensive, low input grazing can be coupled with low emission intensity. Water followed a different trend that was more influenced by regional differences (see section 10.2), though it is
noted that irrigation water was higher in the QLD RAF analysis and this followed a different trend to other impacts for the region. This is possible because small numbers of farms using irrigation can have a disproportionately larger effect on water use than on productivity or other impacts.

The comparison of intensive grain fed beef production and grass-fed production provided a clear demonstration of the role and impact of intensification. Grain finished domestic or mid fed (export) beef utilised 1.2-1.4 times more energy, 4-6 times more crop land, 1.5 times more water and a rapid increase in HEPR, providing a clear trend of higher resources for this system. In contrast, total land requirement was 20-23% lower and LU and dLUC emissions were similarly lower, though method choices and region had a larger effect on estimated impacts than production system. Emission intensity (excl. LU and dLUC) was 16-24% lower compared to market equivalent NSW RAF results. This was slightly different to the results indicated in chapter 4, because lower feedlot manure emission factors have now been confirmed with additional Australian research (see section 10.4.4.1), and therefore are more accurate than the current inventory method. When regional impacts of water stress were removed, water stress was in the order of 10 - 20% higher for grain-fed cattle (as shown by results for the long-fed scenario, which was located in the same WSI region as the grass-fed cattle). This result suggested that substantial increases in water volume may not correspond to large water impacts, and that regional factors are very sensitive.

Grain finishing was found to reduce GHG emissions and land occupation in a similar way to intensive grazing, though more pronounced, with concomitant increases in energy, water and crop land, and the divergence between the systems tended to be more substantial and more consistent (i.e. no ‘low energy’ grain feeding systems were found). However, while the trend in production intensity is clear with grain finishing in this comparison, there are other considerations that should be taken into account. Grain finished beef reduced total land occupation, and associated reductions in land related impacts were not studied in detail in the published studies but could be significant. Potentially large reductions in LU and dLUC, biodiversity and soil impacts may result from reducing grazing land requirements via grain finishing,
though testing these questions would require consequential LCA modelling to sufficiently understand the direct and indirect impacts.

The impact of grain finishing may also not be consistent when seasonal effects are taken into account. Cattle numbers in feedlots increase during some droughts (for example, 2013 and 2015, see BOM, 2017; MLA, 2017) and this may have a disproportionately large benefit for land condition by allowing producers to reduce stocking rates by selling cattle to feedlots in drought years. It is unclear what the energy and GHG emission impacts associated with grass-fed beef during drought would be, but it can be expected that increased supplementary feed, and poor animal performance, will rapidly cause energy demand and GHG emission intensity to increase. Thus, fair comparison of these systems and their role in livestock sustainability must consider drought years and ‘normal’ rainfall years. This revealed a knowledge gap and limitation in the methods used for performing regional assessments, which averaged data over multiple years thus reducing (rather than assessing) the impact of low or high rainfall years. This analysis remains a gap in the research.

10.3.3 SUMMARY OF KNOWLEDGE: PRODUCTION SYSTEM INTENSITY

More intensive production systems did not have a uniform effect on environmental impacts or resource use, though broad trends were evident. Increased production intensity typically resulted in higher energy demand across all meat production systems, though exceptions such as outdoor pork production, which had higher energy demand than intensive pork, were noted. Impacts on water corresponded less with production system and more with region, though in some cases increased intensity corresponded to higher water inputs (for example, irrigation associated with VIC RAF lambs, and water use for flushing in conventional pork). This did not necessarily correspond to higher water impacts in the examples cited because of regional factors (see section 10.2), but it is expected that within the same region (equivalent WSI) higher water impacts would correspond to higher water use.
Production system intensity had a variable impact on GHG emissions. Lower production intensity pork and chicken meat systems had lower emission intensity, while in most cases, higher production intensity for lamb and beef had lower emission intensity. Multiple factors influenced this outcome. In pork production, emissions were dominated by manure and therefore, changes in manure management have a dominant effect on emission intensity. This masked the differences in the impact of lower productivity in the case of outdoor pork. For deep litter pork and FR chicken meat, animal productivity was similar (i.e. comparative FCRs) for the less intensive production systems but energy demand for housing was lower, corresponding to lower overall energy demand. In these systems the impact of feed generally dominated other factors. Because FCRs were comparable, the small differences in housing controlled the overall outcome when regional influences on feed supply were removed from consideration.

Ruminant systems showed a general trend towards lower emission intensity with higher production intensity, and this was particularly apparent with grain finished beef, and may be even greater if seasons/impacts were taken into account. Intensive grazing systems tended to have low GHG emissions per kilogram of product because these systems were associated with high livestock growth rates and high reproductive performance. However, some sheep flocks located in extensive grazing areas were observed that also had low GHG emissions per kilogram of product, because lamb growth rates and reproductive performance was high in these flocks. In these cases, extensive systems often had the same emission intensity as intensive systems, with low energy and crop land utilisation.

Clearly, it is difficult to establish consistent trends from the results presented in this thesis. This contrasted with the theoretical calculations published by some researchers such as Capper (2012) who reported lower impacts across all categories when comparing intensive beef production in the USA compared to grass finishing. Similarly, the same researcher reported that the trend in environmental impacts and resources had declined over time in the USA because of intensification (Capper, 2011). By contrast, Wiedemann et al. (2015a) showed that GHG emissions from the Australian beef slaughter herd declined over time, but energy demand increased substantially, because of intensification. Thus, research must be ‘anchored’ with...
actual data, using a combination of CSFs and regional datasets to be confident that the results reflect industry practice.

To provide a graphical summary of the different effects of intensification across the industries, Table 7 shows the general trends observed in the published studies, with the impact of regions removed to the extent possible.

Table 7. Comparison of impacts from intensive production systems and lower intensity production systems, with regional influences discounted, on resource use (energy, water, crop land, HEPR) and environmental impacts (GHG, LU, dLUC, water stress)

<table>
<thead>
<tr>
<th>Species (meat type) and production system compared</th>
<th>Energy</th>
<th>Water use</th>
<th>Water stress</th>
<th>Crop land</th>
<th>GHG</th>
<th>LU, dLUC</th>
<th>HEPR</th>
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<tr>
<td>Pork (conventional or conventional+deep litter)</td>
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<td>Pork (conventional or outdoor+deep litter)</td>
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<td>Pork (conventional+biogas or conventional+DL)</td>
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<td>chicken meat (conventional or free range)</td>
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<td>Lamb (intensive or extensive grazing)</td>
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<td>Beef (grain or grass finishing)</td>
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<td>Beef (intensive or extensive grazing)</td>
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Notes: Water use is defined as fresh water consumption. HEPR is human edible protein required. LU, dLUC Land use and direct land use change GHG emissions. Red indicates an increase in impacts with higher intensity, yellow is ‘no change’ and green is a reduction in impacts.

Considering these results, it was clear that systems leading to lower GHG emission impacts did not uniformly reduce other impacts. Considering the proliferation of GHG emission research and the prevalence of single indicator LCA research focused on GHG emissions only (McClelland et al., 2018), there is a risk of GHG emissions being seen as a proxy for environmental impact, which is clearly not the case based on the above analysis.

This analysis has considered the impact of production system and production intensity on resource use and the environmental impacts in focus. However, further investigations covering water quality, soil health and biodiversity are required to provide a more comprehensive understanding of this issue in Australia. The results show that the proportion of different production systems can be important for...
defining ‘average’ results, and future research should consider this when aiming to study representative production in a region or nation.

Additionally, the impact of intensification may differ between seasons in some instances (for example grain finishing with beef), and further research is required to understand these interactions.

10.4 BENCHMARKING AND HOTSPOT ANALYSIS

10.4.1 ENERGY DEMAND

Competition for scarce resources, cost pressures and government initiatives to reduce energy demand mean all livestock production systems need to reduce energy demand relative to output to remain competitive. Improvement relies on clear benchmarks and identification of hotspots throughout the supply chain, particularly to guide research and development funding. These studies were the first detailed LCA livestock CSF data of energy demand in Australia, with the exception of two farms in Peters et al. (2010a) that did not have limitations associated with system boundaries. The results showed a similar pattern with respect to energy demand between meat products in Australia compared to previous studies covering the same meat products (i.e. Williams et al., 2006) and the global review of de Vries and de Boer (2010). Where the national study of Wiedemann et al. (2015a) was used to account for energy demand in beef, the trend was beef>pork>chicken meat>lamb. Energy demand for beef (see chapter 3 and chapter 4) was lower than most studies reviewed by de Vries et al. (2015), taking into account the different reference flows, indicating beef from some regions has relative low-energy demand when benchmarked against global production.

Hotspots associated with energy demand were similar for chicken and pork production, but quite different for beef and lamb. Energy demand for chicken and pork was primarily associated with feed grain production and milling, contributing 53 - 59% for chicken meat and 63% for pork retail cuts respectively (see chapter 8 and chapter 9.2). Direct energy demand for operations associated with housing and
animal husbandry were 18 - 21% for chicken meat and 23% for pork respectively, and the predominant energy sources were electricity and gas. Contributions were similar to recent European studies (i.e. Dourmad et al., 2014; Leinonen et al., 2012a), but these did not include impacts from meat processing, which were found to contribute 13 - 16% for chicken meat and 14% for pork in the present studies. Considering the similar supply chain energy profile, this suggests both industries may benefit from similar energy efficiency strategies focused on electricity and gas use, which are increasing costs in Australia.

The lamb and beef studies (chapters 3, 5 & 6) showed a larger proportion (36 - 47%) of energy demand was associated with diesel for farm operations and varied with the level of production intensity as discussed previously. Indirect energy associated with feed production (fertiliser, supplementary feed) contributed smaller proportions, ranging from 15 - 28% for lamb production and slightly lower proportions for grass-fed beef, which was the major reason why these systems were less energy intensive.

Energy demand for meat processing contributed 30% for beef and 31% for lamb respectively. In absolute terms, energy per kilogram of chicken and pork processed was lower than beef and lamb. This was partly explained by the higher proportion of red meat processing plants that included rendering, which is highly energy intensive, while pork and meat chicken processing plants more frequently sent rendering material off-site for processing. This indicates a small inconsistency in the system boundaries between the studies. A sensitivity analysis of this aspect revealed a potential 5% over-estimate in energy demand for beef and lamb compared to chicken and pork respectively.

10.4.2 FRESH WATER CONSUMPTION AND STRESS
WEIGHTED WATER USE

The fresh water consumption and water stress assessment presented in chapters 3 to 9 was the first completed analysis globally that used LCA-equivalent methods, and provided unique insights into livestock production and water use in a water-scarce country. Livestock have been implicated because of the apparent high contribution to
water use, but these assessments have been done at a very high level and have included rainwater used to grow pasture (i.e. Hoekstra and Chapagain, 2007; Pimentel et al., 1997) resulting in very high apparent water ‘use’. However, a key omission in these studies is the lack of assessment of the impact of using water. The studies in this thesis advanced knowledge regarding water use by determining fresh water consumption and applying impact assessment methods.

Trends in water stress impacts could not be compared between species because of the lack of regionalised data for beef, lamb and chicken. Trends in water inventories were similar to the global volumetric water footprint assessments of blue water (Mekonnen and Hoekstra, 2012) who reported a weighted, global average blue water trend of beef>lamb>pork>chicken meat. However, comparing the averaged lamb results in the present studies with the national average for pork suggested that lamb may have higher water use, but lower water stress impacts than pork. Further investigation is required to understand water impacts across all species and regions, and variation between years and within years should be taken into account.

The source of water use for ruminants in the present studies was different to global assessments. Mekonnen and Hoekstra (2012), and the reviews of Legesse et al. (2017b) and the author and colleagues (Doreau et al., 2012) showed that global water use for ruminants was dominated by irrigation water. However, Australian studies showed irrigation was 30% of water use for national average beef (Wiedemann et al., 2015a) while drinking water and supply losses were much greater. Similarly to global averages, irrigation associated with feed grain production contributed 63% of water to pork production (national average) and was the dominant driver of water stress. Taking regional knowledge gaps into account, irrigation is expected to be a substantial driver of water impacts from chicken meat also. Approximated impacts using the values for QLD, SA and NSW diets in chapter 8 suggest irrigation could be >70% of water stress impacts in Australia. The proportion of irrigation water use was higher for export lamb (53%, chapter 5 and 6) than Merino lamb (chapter 7, and supplementary material). Taking knowledge gaps with respect to regional production into account, it is likely that irrigation water use is lower than estimated for export lamb when the impact of lambs from Merino flocks is fully accounted for.
Water used directly for drinking, cleaning and cooling purposes represented 8-33% of total water use in the included studies, and was highest for beef cattle both in proportion and in volume, because of the relatively high ambient temperatures experienced in Australian grazing regions and the effect of this on drinking water (CSIRO, 2007). Fresh water consumption results in the present studies were more comprehensive than previous Australian beef and lamb studies, resulting in higher levels of fresh water use for similar regions and systems than the Australian literature (see Table 4 and Table 5). In the majority of grazing regions, where irrigation use is minimal, supply losses were the largest water use for ruminant livestock. These were predominantly evaporation losses from farm dams, which are known to be a major contributor to losses in the Murray Darling Basin (see Nathan and Lowe, 2012). The methods applied in chapter 3 were considered a critical advance in the knowledge of supply losses, because this was the first study to conduct farm dam water balance modelling using historic climate data and farmer knowledge to verify dam water levels and evaporation rates using supply system water balances. On-farm dam inventory data was verified using GIS to confirm the number of dams and the surface area, ensuring that key factors in the water balance were correct, relative to livestock numbers on the farm. In total, water balances were modelled and verified against farmer records for >1000 water supply sources in every major state of Australia, providing a sound starting point for understanding these losses. With the exception of Ridoutt et al. (Ridoutt et al., 2012a; Ridoutt et al., 2012b) who provided a conceptual model without farm inventory data, previous research had overlooked these losses or omitted them completely from the analysis (i.e. Harding et al., 2017; Legesse et al., 2017a; Mekonnen and Hoekstra, 2012; Palhares, 2017; Ran et al., 2016). Water supply losses also occur with irrigation systems, and these were reported to be 27.1% of irrigation water supplied in Australia for the years 2004/05 to 2010 (ABS, 2006; ABS, 2012)

These losses are clearly significant in Australia and further consideration of these impacts is required in other regions of the world that utilise surface water storages.
10.4.3 LAND OCCUPATION AND GRAIN UTILISATION

10.4.3.1 LAND OCCUPATION

The studies presented in this thesis provide some of the first LCA data to distinguish between crop land and rangelands (non-arable land used for pasture) across multiple meat production systems, as recommended by de Vries and de Boer (2010). Broad observations of total land occupation showed similar trends between species to those found in the international literature (de Vries and de Boer, 2010). Total land occupation was higher for all Australian meat products than the northern hemisphere studies reviewed by de Vries and de Boer (2010). This reflected the difference in productivity between Australia and many northern hemisphere regions and particularly, Europe. Similarly, Australian cereal grain production requires more land to produce the equivalent grain production to European systems, as Australian yields are typically less than 2t per hectare (ABS, 2013), compared to 5-8t per hectare in western Europe (Lin and Huybers, 2012).

Broad trends in crop land occupation between the different meat production systems were more significant because crop land is a limited resource in Australia and globally (ABARES, 2017; FAO, 2009). Noting the regional and production system impacts and knowledge gaps, crop land occupation followed the trend: pork > chicken meat > lamb in the published studies. The analysis of beef was influenced by the regions assessed and has not been assessed at the national level. National impacts may be higher than reported for grass-fed beef in chapter 3 and 6, though this depends on a combination of factors. These include the contrasting impacts of southern beef regions (likely higher crop land), northern pastoral regions (lower crop land), and grain fed beef enterprises (much higher crop land). Further research is required to understand this in detail, and this could be advanced by utilising spatial land resource data.

Grass-fed lamb and beef utilise large areas of rangelands in Australia that have fewer competitive uses than crop land (ABARES, 2017). From an LCA perspective, it is important to note that rangeland occupation cannot be considered from an efficiency perspective only. Poorer outcomes can result from ‘reduced’ land occupation where
this is achieved with higher stocking rates, because high stocking rates can result in land degradation and loss of biodiversity (Ash et al., 1997; McIntyre et al., 2003). This is a clear case where trade-offs need to be considered, taking into account a broader range of impacts than assessed here. For example, those associated with soil health and water quality are highly relevant to understanding grazing impacts in rangelands, as are biodiversity impacts. However, the latter has proven difficult to integrate into LCA, and future studies are required to adapt and apply methods in the Australian context (Moore, 2015). Considering the very large land areas covered by Australia’s grazing and cropping regions, and the advances that have been made in satellite imagery and remote sensing, there is considerable scope for integrating LCA and GIS to advance knowledge in this area (Moore, 2015). With advances in the capability of ‘big data’, researchers in the future may be able to integrate detailed livestock information such as location and growth rates with spatial data, such as land cover (Geoscience Australia, 2010), opening new possibilities for integrated research.

Another area requiring further research relates to the role of livestock in mixed farming regions. One conclusion that could be extended from the current research is that utilising arable land for ruminants should be minimised to the greatest extent possible. However, the role of grazing livestock in mixed farming regions requires further consideration in LCA before drawing such a conclusion. Eady et al. (2012) studied a cropping and sheep farm and proposed methods for examining the impact and contribution of each system to the other, making an important contribution to the subject. Additionally, the role of legume pasture leys (and therefore livestock) on soil and crop health and nutrition (Chan et al., 2011; Holford et al., 1998) should be investigated using LCA. For example, improved knowledge regarding the impact of pasture leys on soil carbon flux over long time horizons is required, and livestock emissions and impacts need to be integrated into this to provide a comprehensive understanding of the benefits and trade-offs.
10.4.3.2 Grain Utilisation

Livestock systems are a substantial contributor to human nutrition globally, with 40% of protein and 18% of energy being supplied by livestock systems (FAO, 2017). However, a high reliance on grain in some livestock systems can reduce the net contribution of livestock to food supply because of the inefficiency of converting grain protein to meat protein. Assuming continued population growth, it is foreseeable that competition will increase between grain for human food and grain for feeding livestock, potentially causing conflict with food security. Consequently, HEPR (and similar metrics) are important to track in the future (Capper and Bauman, 2013; Gill et al., 2010; Wilkinson, 2011).

Significant differences in grain utilisation rates were observed between the monogastric and ruminant species in these studies, though this was influenced by production intensity. Australian pork and chicken meat production systems utilised > 4kg of human edible protein in grain per kilogram of meat protein from the system, and these results are not expected to be influenced too heavily by regional or production intensity effects.

The analysis of beef and lamb is more complicated because production intensity has a large influence on results. The lamb systems studied used less grain, though only a small proportion of grain finishing was included in the analysis and this may not be representative. Nonetheless, HEPR was 0.6 kg, which was substantially less than for pork or chicken meat. Grass finished beef had even lower levels of 0.1 kg, but a national average that took grain feeding into account would be much higher. Grain finishing systems reported in chapter 6 (reported these as the inverse, HEP-CE) were 2-3.3, though the upper value is for long-fed beef which is produced in small volumes in Australia. Grain-finished beef is approximately 35% of all beef processed in Australia (MLA, 2016), and average days-on-feed for feedlot cattle were found previously to be 114d (see Wiedemann et al. 2014). Consequently, the HEPR for Australian beef may be in the order of 0.7-1.2, which is slightly higher than the estimate for lamb. Further research is needed to test these assumptions and understand the drivers more clearly.
While HEPR is useful for broad comparisons, some caveats are required with respect to the general role of meat in the diet. It is important to note that gross protein and energy are only part of the contribution made by meat in the diet; for example, minerals such as iron are an important contribution to human nutrition made by red meat (McAfee et al., 2010; Pereira and Vicente, 2013; Williams, 2007).

Additionally, meat and grain-based protein have different nutritional properties for human diets because of the different balance of amino acids in each (FAO/WHO, 1985; Wu, 2009). Consequently, ‘human edible’ grain protein is not nutritionally equivalent to animal protein. Considering this, meat is expected to have a higher effective protein content in a human diet than the grain-based equivalent and the HEPR may be slightly over-estimated.

The analysis conducted for this study accounted for the proportion of pork and chicken meat diets that utilised non-human edible diet inputs such as meat and bone, blood meal, feather meal and fish meal in diets. Utilising these by-products reduced the requirement for plant protein sources, and value-added to meat processing by-products not suitable for direct human consumption.

Clearly, future competition for grain resources poses a threat for livestock systems that are heavily reliant on these resources. Further research is required into alternative feed sources for monogastrics such as waste food from the human supply chain, insects and potentially algae. Pigs also have an established, important role in utilising food rejected from the human supply chain pre-consumption and the significance of this role warrants examination. To advance this, a first principles analysis by the author was performed to investigate the role of waste food from the human supply chain as a source of inputs for pork production. Approximately 40% of food in the Australian human supply chain is wasted (FAO, 2013; Gustavsson et al., 2011; Lapidge, 2015), representing some 22 M GJ of food energy (equivalent to 1.23 M tonnes of cereal grain) that is potentially available in Australia annually. With full energy recovery, this corresponds to 78% of the feed requirements for the Australian pig industry (based on feed requirements outlined in 9.1). Because of the regulatory barriers associated with swill feeding, a proportion of this food waste would require treatment such as cooking, or could be used as a feed source for
insects that could be subsequently fed to pigs (Makkar et al., 2014; Veldkamp and Bosch, 2015). Insect FCRs from better species are in the order of 1.9 (Oonincx et al., 2015) and thus total available food inputs (if 100% of food waste was fed to insects) would contribute some 46% of the feed requirements of the Australian pig industry.

This analysis does not account for the challenging logistics of utilising high proportions of food waste. Nonetheless, this does demonstrate the potential of the pig industry to improve the efficiency of the overall food supply chain via consumption of food waste.

Other options for novel feed production systems such as algae have also been investigated by Australian researchers, with some studies considering the potential of utilising effluent from pig production (Murdoch University, 2015) or other waste streams as a feedstock for algae production. While no operational systems have been identified yet, this novel solution offers the potential for new, non-competitive feed sources produced from low value by-products, though competition may also exist for using this as a feedstock for bioenergy (Bharathiraja et al., 2015).

Competition for arable land may also become an acute problem for ruminant species in the future. Research into high yielding, high energy pasture species and pasture systems is an important, ongoing requirement for future production, as is development of a better understanding of the role of livestock in mixed grazing systems where pasture leys may have benefits for soil health and crop production.

10.4.4 GREENHOUSE GAS EMISSIONS

Results presented here show that total GHG emissions, gases and sources varied markedly between regions and production systems. Because national analyses have not been performed for lamb and chicken, a clear trend is not known. However, it is expected that impacts between species will follow the same trend observed globally (de Vries and de Boer, 2010), though it is presented here inclusive of lamb: beef>lamb>pork>chicken meat.
Total GHG emissions from cradle-to-gate beef and lamb is driven by the large contribution from enteric methane; 83-90% for grass-fed beef and 79-86% for lamb with LU and dLUC excluded (chapter 3-7), while nitrous oxide and carbon dioxide from energy are smaller contributors. Meat chicken production was dominated by carbon dioxide from energy demand (> 70% excl. LU and dLUC), followed by nitrous oxide (approx. 20%), which arose from the cropping phase and from manure. In contrast, emissions from average Australian pork (excl. LU and dLUC) were dominated by methane from manure treatment (approx. 58%) and carbon dioxide from energy (33%) with smaller amounts of nitrous oxide. Manure emissions from grazing ruminants were less significant than for pork or feedlot beef, because manure is deposited directly to pasture where the small application rates and generally fast drying rates are not conducive to the emission of methane or nitrous oxide. Manure emissions from chicken meat were relatively low in comparison to pork and feedlot beef, because of the low direct emissions from manure deposited on litter.

Impacts from LU and dLUC were dominated by regional factors and sufficient data were not available to understand them at the national level. However, they can clearly be high for beef in some regions such as QLD. Considering the results for southern beef, lamb and wool production, these high impacts are expected to be counterbalanced by lower ones in southern Australia. Further research is needed to quantify these regional influences. Inclusion of LU and dLUC impacts in lamb and wool assessments had minimal effects on total emissions, and potentially resulted in a decrease in emissions because of modest rates of carbon sequestration in pastures. Similarly, LU and dLUC impacts were minimal for pork (6% increase) and these were predominantly associated with emissions from imported soymeal. In contrast to these results, LU and dLUC emissions increased GHG emissions impacts from chicken meat by a factor of 1.2-1.6, with the variation caused by different inclusion rates of soymeal, which contributed 94% of the LU and dLUC impacts for chicken meat.

The study presented in chapter 3 quantified the influence of beef herd productivity factors on total emissions, showing that weaning percent, age and average daily gain (ADG) could explain 0.33 and 0.47 of the variability in GHG emissions according to the regression equation:
GHG = 20.73 – 0.047Wean% - 11.06ADG + 0.13CP
R² = 0.87

However, this model is limited by the size of the dataset and the regions covered, and emissions were under-predicted for herds producing live export cattle in northern Australia (Wiedemann et al., 2015d). Considering Eady et al. (2011) showed higher emissions for weaner steers than slaughter weight steers, the difference in liveweight between live export steers (typically 300-340 kg LW) and grass finished slaughter weight steers (mean: 424-597 kg LW, chapter 3) may explain part of this. This finding suggested a larger dataset (or regionally specific datasets) would be required to develop a robust prediction equation for total GHG emissions from Australian beef herds.

10.4.4.1 Sensitivity Analysis

Greenhouse gas emission estimation is constrained by the large number of emission categories and the dearth of research on some emission sources.

The initial studies (presented in section 2.5) identified specific, sensitive manure management estimation techniques and emission sources that had a large influence on whole of system emissions. This included sensitive emission factors for predicting manure nitrous oxide, methane and ammonia from grain-fed beef systems, chicken meat production and pork production. Following the initial LCA research, a range of recommended improvements to manure estimation and GHG methods were made, and these were summarised in a review by the author (Wiedemann et al., 2014c). Subsequently, these revised methods were incorporated into the NIR (Commonwealth of Australia, 2016) to improve manure GHG emission estimation for feedlot beef cattle, pigs and poultry.

Following this revision, new research has been released in Australia and globally, increasing the knowledge base regarding GHG emissions from manure. This was taken into account by performing additional sensitivity analyses for grain-fed beef (see chapter 4) and chicken meat (see chapter 8). This revealed further points of
discrepancy between the scientific literature and the Australian NIR. A summary of the sensitivity analysis is provided in Table 8, providing recommendations and supporting literature to guide future LCA and inventory method revisions for the NIR.

**Table 8. Sensitivity analysis testing emission factors and assumptions relating to the estimation of manure greenhouse gas emissions, with recommendations for research and modification of inventory methods**

<table>
<thead>
<tr>
<th>Source and Emission factor</th>
<th>Current NIR factor and supporting science</th>
<th>Alternative proposed factor, sensitivity and supporting science</th>
<th>Recommendation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Chicken meat – nitrous oxide from manure, emitted during housing</td>
<td>0.001 kg N₂O-N / kg N excreted based in IPCC (Dong et al., 2006), based on expert judgement (no cited research).</td>
<td>0.0035 kg N₂O-N / kg N excreted Increases manure GHG emissions from grow-out stage by 206% and total supply chain emissions by 3% Based on measured emissions from four shed trials under Australian conditions (Wiedemann et al., 2016d)</td>
<td>The research trial provided the first data available for Australian conditions. The literature cited by Wiedemann et al. (2016d) and more recent research (i.e. Pereira, 2017) typically show higher emissions than the IPCC factor indicates. Application of experimental data is preferred to ‘expert judgement’, supporting application of the higher value based on Australian research. Revision of the NIR factor should be considered during the next review.</td>
</tr>
<tr>
<td>Chicken meat – ammonia from manure, emitted during housing</td>
<td>0.3 kg NH₃-N / kg N excreted Based on a review of international literature completed for the Australian National Pollutant Inventory (no Australian research).</td>
<td>0.11 kg NH₃-N / kg N excreted Manure GHG emissions varied by 8% in response to this factor, and total GHG emissions by &lt;1%. Based on measured emissions from four shed trials under Australian conditions (Wiedemann et al., 2016d)</td>
<td>The research trial provided the first data available for Australian conditions. This value tended to be lower than other studies. Further research is required to confirm a much lower factor for Australian conditions and in the interim. The current NIR value is supported.</td>
</tr>
<tr>
<td>Feedlot – nitrous oxide from pen surfaces</td>
<td>0.02 kg N₂O-N / kg N excreted</td>
<td>0.0-0.006 kg N₂O-N / kg N excreted, average 0.0024. 70% decrease in manure emissions 16-19% decrease in feedlot gate to gate emissions</td>
<td>Revision supported by other recent Australian research (Bai et al., 2015; Bai et al., 2016; Denmead et al., 2014; Redding et al., 2015b; Sun et al., 2016) Revision of the NIR factor should be considered during the next review.</td>
</tr>
</tbody>
</table>

Table 8 indicates two sensitive factors that were significant for the prediction of manure management emissions. Revision of the nitrous oxide emission factor for meat chickens is expected to result in an increase in predicted meat chicken manure emissions of 206% (as calculated in the LCA research – chapter 8) while revision of the nitrous oxide factor for feedlot pen surfaces is expected to result in a 70% decrease in manure emissions. However, feedlot manure emissions are a much larger source than meat chickens in both LCA research and in the NIR, thus the impact of the second factor is much more significant, resulting in a 16-19% (feedlot gate-to-
gate) reduction in GHG emissions. An objective of this thesis was to identify sensitive and uncertain emission factors and provide recommendations for review and potentially revision of these factors for application in LCA research and the NIR. This sensitivity analysis highlights the significance of two factors where changes are recommended.

10.4.5 SUMMARY OF KNOWLEDGE: BENCHMARKING AND HOTSPOTS

The studies in this thesis provided a substantial livestock energy, water, GHG emissions and arable land benchmarking dataset for Australian conditions. The trend in energy demand, GHG emissions and fresh water consumption between species was similar to the observed global trends, though water impacts showed a potentially different trend, with water stress impacts for lamb potentially being lower than pork. Insufficient data were available to understand these for Australian beef. In contrast to the global literature, supply losses were a significant source of water use, and in some regions water stress, for ruminant species.

Land occupation for Australian meat production was higher than northern hemisphere studies because of the lower relative yields of pasture and crops in Australian agriculture, and ruminants utilised considerably larger areas of land than monogastric species. However, crop land occupation, which is a more constrained resource in Australia, was higher for monogastric species compared to lamb and impacts were less certain for beef because national averages have not been determined. This analysis was more advanced than previous comparisons between the species because of the separation of crop land and rangeland and thus contributes to the global literature by addressing this knowledge deficit. Similarly, HEPR was higher for the monogastric species, which are reliant on grain, and quite low in comparison for lamb and grass-fed cattle, though average values for beef, accounting for grain finishing, were higher.

Inclusion of LU and dLUC emissions increased emissions for chicken meat, pork and beef, and regional influences were very pronounced. Lamb production, in the studies
included here, was less influenced by LU and dLUC, though methods are uncertain and impacts are regionally variable, meaning these results should be viewed with caution.

Considering the variability observed between regions, production systems and levels of production intensity, further research and in the case of chicken meat and lamb, national analyses, would be beneficial to understand and expand the knowledge base for all resource areas and impact categories. Utilising this knowledge regarding impact hotspots, an investigation of GHG emission mitigation opportunities and associated trade-offs is included in the following section.

10.5 MITIGATION AND TRADE-OFFS

Mitigation research in Australia has focused on single impact categories (for example, GHG emissions) and has rarely considered trade-offs with other impacts by applying LCA methods. While the present analysis is only partial (for example, impacts associated with water quality, land health and biodiversity were excluded) it still provides some insights with respect to mitigating GHG emissions and resources, and the trade-offs that may exist between these, addressing a key objective of the thesis.

The previous section elaborated on the total impacts and hotspots associated with chicken meat, pork, beef and lamb production. Here, mitigation is discussed primarily within the context of each meat production system, but brief consideration is also given to preferential selection of one meat compared to another. Mitigation options were grouped into categories of similar mitigation pathways, reflecting major hotspots (section 10.4) where practical opportunities were found to reduce impacts. These were exemplified with case studies from the thesis chapters, acknowledging the analysis was not comprehensive. Recommendations for future mitigation research are provided in section 10.6.
10.5.1 MANURE MANAGEMENT AND UTILISATION – PORK AND CHICKEN MEAT

The pork and chicken meat industries generate large amounts of manure from confined animals, which is a substantial source of GHG emissions (particularly for pork production). In conventional pork production, water is also required to transport manure from the sheds. The pork industry is in a unique position to reduce GHG emissions because the largest emission source from conventional production is the methane released from effluent ponds. Methane has a high energy potential and can be captured to generate heat or electricity. Piggeries that capture pond methane are also eligible for carbon credits under the Government’s ERF, resulting in favourable economics for larger piggeries (> 1000 sows, farrow-finish) (McGahan et al., 2013). Some 13% of the industry has installed biogas capture systems, with all but one having been installed since 2010 (Skerman and Tait, 2016). The analysis presented in chapter 9.1 showed that pond-covering and energy generation reduced GHG emissions by 60 - 64%, while energy demand was reduced by 8 - 25%. Few known trade-offs exist with this system. Nutrient treatment, water and land occupation remained the same, indicating this is an ideal mitigation option. Further to this, biogas and energy generation often results in surplus energy at piggeries (particularly heat) which opens the prospect of utilising this energy to improve the efficiency of water utilisation from effluent (via water treatment) and improving the utilisation and recovery of nutrients for fertiliser (Murphy et al., 2016). This is an important future direction for the industry and may enable pork production to ‘close the loop’ with respect to energy, water and nutrients, which would represent a major step forward for environmental management and efficiency in the industry.

Opportunities also exist to reduce water by improving utilisation of piggery effluent to replace irrigation water supplies. This was found to reduce piggery water use by 18% (section 9.2) and this could be realised if piggeries were co-located with irrigated crop farms.

Considering the opportunities demonstrated for pork production, further analysis was completed for the chicken meat industry using the production systems reported in chapter 8. Manure from this system is handled as a dry litter (manure mixed with
bedding materials such as pine shavings, sawdust or straw) and therefore has very different properties to pig manure. Analysis by the author (Wiedemann et al., 2012b) demonstrated that anaerobic digestion (AD) or combustion were able to reduce GHG emissions by 25-32% and energy demand by 20-27% per kilogram of chicken meat produced by reducing direct manure emissions and the demand for fossil fuel energy sources. As with the pork industry example, water and land occupation were not changed, suggesting low risks of trade-offs. This technology has not been adopted by the chicken meat industry to date, and further research and development is required to improve adoption.

A further mitigation option associated with chicken meat, pork or feedlot production systems is via improved manure nutrient utilisation as fertiliser. Urea, a common nitrogen fertiliser, is GHG emissions and energy intensive to produce (1.9kg CO₂-e and 54MJ per kg N, Ecoinvent, 2018) and reducing fertiliser requirements by utilising poultry, pig or feedlot manure can therefore offset small amounts of GHG emissions and energy with little effect on water use or land occupation. Modelling by the author showed a reduction in energy demand of 6 - 10% for meat chicken production (Wiedemann et al., 2012b) and 1% for pork production (Wiedemann et al., 2012a). This mitigation could be improved if the plant nutrient availability of manure could be improved, which is an area of active research for both industries. Carbon sequestration can also be achieved via manure application to crop and pasture land (Wynn et al., 2006) though researchers have found this difficult to quantify under Australian conditions (M. Redding pers. comm.).

10.5.2 IMPROVED FEED CONVERSION RATIO

In addition to reducing manure related emissions, improved FCR can decrease GHG emissions and energy demand for pigs and poultry, if all other factors remain equal. The effect of a 0.1 change in FCR was found to result in 3 – 5% reductions in GHG emissions, energy, arable land and fresh water consumption for chicken meat (chapter 8), but gains must be made without changing factors that influence impacts. This may be possible via better animal husbandry or genetic gain over time. In pork production, FCR was found to explain 88% of the variability in GHG emissions for
conventional production systems. As well, variability in GHG emissions between farms with lower FCRs (2.4) and higher FCRs (3.8) was 40%, suggesting improvements may be possible at some piggeries via management practices that reduce FCR, provided other factors such as diet composition, shed type or shed energy inputs are not affected.

10.5.3 DIET MODIFICATION - BEEF

Mitigation of GHG emissions from beef production is a topic of importance for the Australian industry, because livestock emissions are very high relative to other meat production systems. To examine the potential trade-offs in mitigation of emissions from beef, an additional mitigation analysis was included, based on Wiedemann et al. (2013). Assumptions are provided in Appendix 2. This analysis investigated more rapid growth rates to reduce life time feed intake and enteric methane emissions.

The analysis compared mitigation scenarios to a baseline management scenario, which was to finish steers on native pastures to export weights (low input grass finishing, 600kg LW). The mitigation scenarios were as follows:

- Forage plus feedlot (75d)
- Leucaena plus feedlot (75d)
- Leucaena with revegetation (fast growth rate backgrounding and finishing on leucaena, with a proportion of land set aside to allow woody regrowth acting as a source of carbon sequestration).
- Low input grass finished
- Forage plus feedlot (120d)
- Grass, forage and supplementary feed
- Irrigated Leucaena and supplementation

Results (see Figure 2) showed an emission reduction of 17-41% compared to a baseline management scenario. GHG emissions reductions of 29-31% were achieved with high growth rate irrigated leucaena scenarios. The forage + feedlot scenarios (either 75 or 120d) resulted in a 22% and 25% reduction in GHG emissions, while forage and supplementary feeding resulted in a 17% reduction in GHG emissions.
When the impacts of GHG emissions fluxes from vegetation and soil were taken into account in leucaena, the total mitigation increased to between 20-41\% (see Figure 2). In most cases mitigation was achieved by utilising other resources. The highest level of mitigation was achieved by feeding cattle on irrigated leucaena, which utilised a considerable amount of water (5570L kg LW$^{-1}$) and arable land (12m$^2$ kg LW$^{-1}$) compared to the baseline. Grain finishing required higher amounts of energy, grain and arable land, corresponding with the findings of chapter 4.

These scenarios revealed that improved productivity achieved via diet modification typically results in a trade-off with resource use. These results provide a contrast to the pork mitigations which were able to demonstrate reduced GHG emissions with few trade-offs.
Figure 2. Improvement scenarios for backgrounding and finishing steers showing impacts on energy demand, stress weighted water use, greenhouse gas, net greenhouse gas, arable land occupation and grain use compared to QLD CSF export cattle (low input grass finished)
10.5.4 SOIL AND VEGETATION CARBON SEQUESTRATION ON BEEF AND LAMB FARMS

Utilising carbon sequestration to mitigate GHG emissions on sheep farms may also offer opportunities, as shown in the analysis of wool farms in chapter 7. Tree planting for shelter belts was able to reduce livestock related emissions by approximately 10% over a 20-30 year period depending on assumptions for tree growth rates. However, the impact of shelter belts on production and land occupation was not investigated in this analysis, and results from studies reviewed in the literature vary. Bird et al. (1992) suggested that strategic tree planting in southern Australia may increase crop, pasture and livestock production primarily because of a combination of micro-climatic effects and long term benefits associated with nutrient cycling, erosion reduction and dryland salinity management. In contrast, field scale experiments such as Sanford et al. (2003), show reductions in pasture production and therefore carrying capacity in response to competition between pasture and tree belts. Carbon sequestration also only occurs for a defined period of time, while trees are actively growing, and therefore cannot provide a permanent offset for livestock emissions. The implications of the trade-offs between soil carbon sequestration in trees and carrying capacity requires further research. For example, dedicating land to sequestration and therefore increasing stocking rates on remaining land may result in negative impacts on land condition. Increased tree cover can also influence hydrological systems and cause changes to water flows in river catchments, resulting in a significant trade-off with fresh water and water stress (Wiedemann, 2014). Alternatively, reducing stocking rates, and total beef or lamb production, may result in an increase in demand for these products from other production systems globally, potentially resulting in higher emissions. For example, Cederberg et al. (2011) showed that emissions from Brazilian beef, produced on land that was recently deforested, was more than 700 kg CO$_2$-e / kg CW. Further research using consequential LCA is required to understand these potential market based trade-offs.

Improved rates of carbon sequestration under pastures were found to result in emission reductions of up to 20% of lamb emissions, based on the soil sequestration rates of Chan et al. (2010). However, these high rates of sequestration have not been shown in other studies (i.e. Wilson et al., 2011; Wilson and Lonergan, 2013). Higher
soil carbon has been found to be positively correlated with increased soil phosphorus and nitrogen (Wilson and Lonergan, 2013) suggesting that fertiliser inputs (with associated embedded emissions) may be required to achieve improvements, resulting in higher energy demand. However, detailed case studies are required to explore these interactions further.

10.5.5 MODIFIED WATER SUPPLY SYSTEMS - BEEF AND LAMB

Water supply system losses in sheep and beef systems represent substantial losses at the national scale and reducing these offers the most substantial opportunity for lessening water use impacts in these systems. The author has elsewhere demonstrated a reduction in water losses in the Australian beef industry historically (Wiedemann et al., 2015a), with most of this improvement being the result of controlling the flow of artesian bore water. Reducing losses from the other major source of water loss (evaporation from farm dams) is less feasible. While technologies such as covers are available to reduce surface evaporation, costs are typically too high at the current time to utilise these technologies and they may present management difficulties. Water supply efficiency can also be altered by changing the volume of water stored (measured at the catchment level in ML storage / km² – Nathan and Lowe, 2012) (measured at the catchment level in ML storage / km² – Nathan e Lowe, 2012)(measured at the catchment level in ML storage / km² – Nathan and Lowe, 2012) and/or by modifying dam efficiency factors such as the surface area to storage volume. These factors are governed by the number of dams and the design of those dams. Analysis of the CSF datasets showed that where dams are used exclusively, and a dam is located in all or most paddocks, dam density is typically high. Alternatively, farms that had a mix of water supply sources (bore, creek) and farms that had fewer large paddocks were able to utilise fewer dams.

While no studies specifically investigated different water systems to mitigate losses, a useful comparison was found between two NSW CSFs located in the Armidale district (reported in chapter 3). Dam density on these two farms differed substantially (10.8 and 4.6 ML per km²), and the farm with the higher dam density (the
comparison farm) was fairly typical for the region, where a dam is located in the majority of paddocks (unless the paddock had a creek) to provide livestock water. The farm with the lower dam density (mitigation farm) had built fewer, larger dams in some paddocks and had a network of troughs connected to these dams, resulting in a demand factor (extraction relative to total volume) almost 4 times higher than the comparison property (0.11 compared to 0.03) and resulting dam supply efficiency was 32% for the mitigation farm and 17% for the comparison farm. The impact of these differences in dam water supply was 31% lower water demands per kilogram of product for the mitigation farm compared to the comparison farm. These results suggest that using larger dams with reduced evaporation rates, and reticulating water to grazing paddocks can improve efficiencies. Where pumping is required to achieve this, a trade-off will exist with slightly higher energy (and GHG emissions) associated with pumping. Achieving these improvements could be done incrementally by increasing livestock numbers and stocking rates while maintaining the same number of dams.

However, it should be noted that reducing livestock numbers will not result in commensurate reductions in water use unless dams are decommissioned to reduce evaporation (because evaporation is a function of dam density and dam efficiency, independent of livestock drinking requirements). This finding is significant when considering the likely impact of changing the type of meat produced on water use (discussed below). Another approach to reducing water losses is to preferentially utilise water from bores or water courses, to reduce evaporation rates. However, systems requiring water to be pumped may result in higher energy demand (and GHG emissions) for pumping water, and the total energy requirement will be dependent on local topography and the extent that water can be moved by gravity. Fossil energy demand for pumping can be minimised by using solar, which is a popular option.
10.5.6 MEAT PROCESSING

The studies also show that energy efficiency in meat processing is important for overall product efficiency across all species. This represents a hotspot that could benefit from cross-sector research to reduce energy demand. One recent development of significance is the installation of biogas to reduce GHG emissions and offset energy requirements, which will reduce fossil fuel energy demand from this part of the supply chain in the future. This and other opportunities may lead to substantial reductions in future meat processing energy demand (Colley, 2011).

10.5.7 CHANGING MEAT TYPES

Changing between meat producing livestock species has been recommended by some (Foley et al., 2011; Garnett et al., 2017) as a means of mitigating environmental impacts and particularly GHG emissions. However, changing meat types is a substantial change, potentially requiring new production systems and infrastructure, and having a potentially large influence on resource use, including in later stages of the supply chain not covered by the present research. Thus, such an analysis is best performed using consequential LCA.

Having noted this, the research clearly defined trade-off between the species with respect to the utilisation of crop land, utilisation of human edible protein and GHG emissions. As has been extensively documented, GHG emission intensity was high for lamb and beef, and lower for pork and chicken, because of the key differences between the digestive systems of ruminants and monogastrics. The strong contrast between GHG emission intensity and HEPR (Figure 3) highlights this trade-off and key difference between the species. Importantly, this presents a limitation to simple recommendations around changing meat types to mitigate GHG emissions, because each system relies on a different land type. In Australia, crop land is only a small fraction of total land area (Lesslie and Mewett, 2013), limiting the potential expansion of meat types reliant on crop land. In contrast, large areas of rangelands are available and are unsuited to other production systems. Reducing red meat production in rangeland areas to reduce GHG emissions would be expected to reduce
gross meat production for human consumption from Australia, because these production systems cannot be replaced by the other dominant domestic meat production systems. This would reduce Australian exports and therefore global food supply, resulting in changed supply-demand dynamics in other regions of the world.

Figure 3. The relationship between greenhouse gas emissions (bars) and human edible protein required (blue line) for meat produced in four different Australian livestock systems

Notes: Data were developed from available regional and national datasets, but are not representative of average Australian production in all cases. HEPR for average of beef production in Australia has not been determined in the published studies, but will be higher than reported for grass-fed beef because of the influence of grain finishing (Author estimates range from 0.7-1.2). GHG emissions include LU and dLUC emissions. For grass-fed beef this was averaged across QLD (a high dLUC region) and NSW (a low dLUC region).

10.5.8 SUMMARY OF KNOWLEDGE: MITIGATION AND TRADE-OFFS

Investigators of GHG emission mitigation strategies have rarely considered the likelihood of trade-offs with resource use. This review integrated findings from the published studies, highlighting case studies where GHG emission mitigation corresponds to a reduction in resource use (for example, biogas manure management or deep litter housing for pork). It also discovered contrasting instances where lower
GHG emissions were associated with elevated crop land, energy demand and in some instances, water resource inputs, such as with some diet modification strategies for beef. Considering these results, further investigation of the potential for vegetation or soil carbon sequestration, via selective tree planting or revegetation, has merit and may reduce the risk of trade-offs.

There is potential to apply the best mitigations, such as biogas production, from one industry to another. However, detailed feasibility assessments are required, and regulatory barriers must be taken into account. With all the mitigations investigated, results should be considered preliminary, as a more detailed consequential LCA model is required to take into account indirect impacts.

10.6 RESEARCH IMPACT

The research contained in this thesis made an original and significant contribution to the knowledge of environmental impacts of livestock production systems in Australia. To demonstrate this in addition to the discussion, the following section outlines three tangible areas in which the research has had an impact.

10.6.1 IMPACT ON GHG EMISSIONS RESEARCH AND INVENTORY METHODS

As noted in section 2.5, the initial studies were used to identify factors that required revision in the Australian NIR to increase the comprehensiveness and accuracy of the inventory calculations, via the author’s review (Wiedemann et al., 2014c). This resulted in revision of some 95 factors, with around half of these not having been included in the inventory prior to this point.

Further to this, chapters 4 and 8 also demonstrated the sensitivity of two manure emission factors where a further revision to the NIR is recommended.
The hotspot analysis performed in this LCA research was used to direct GHG emission mitigation research in the grain-fed beef, pork and chicken meat industries, resulting in new knowledge and potential avenues for reducing emissions from these industries. Of specific relevance were the results generated for the pig industry, which demonstrate the lower emissions from deep litter production and short HRT effluent treatment systems. To demonstrate the power of LCA research to identify and prioritise effective mitigation strategies, it is noted that, of the mitigation research funded in the $8M National Agricultural Manure Management Program (NAMMP), only the two mitigations noted above (deep litter housing and short HRT manure) were found to be cost effective, and of sufficient merit to be considered for development of ERF methods (see Wiedemann et al., 2016c). Both these mitigation pathways are currently being investigated for incorporation into a revised ERF method for manure emission mitigation. This extends the previous application of the initial pork LCA research, which was used to assist development of Australia’s first mitigation strategy under the CFI (now ERF), the *Destruction of Methane Generated from Manure in Piggeries - 1.1. Methodology Determination* (Commonwealth of Australia, 2015b). Additionally, the author’s research was used effectively in the development of the first emission intensity method for mitigating GHG emissions from beef cattle via improved management of beef cattle herds (Commonwealth of Australia, 2015a), which was extensively reviewed and revised to address leakage risks using the author’s LCA research.

**10.6.2 BENCHMARKING AND EFFICIENCY**

With respect to establishing benchmark targets for industry, the pork LCA research was used to establish the first LCA based benchmarks and targets for mitigating GHG emissions from an Australian livestock industry, via the adoption of formal targets as part of the Australian pork CRC (Pork CRC, 2018). LCA research is currently underway by the author to determine the improvements achieved by the pork CRC in reducing GHG emissions, and this will be used to report on the effectiveness of the research program.
With respect to benchmarking, the energy hotspot analysis from this research has been used to develop detailed energy efficiency research and guidance for the chicken meat and pork industries (McGahan et al., 2014a; McGahan et al., 2014b). In the farm energy review of Chen et al. (2015), the author’s LCA research and associated work was cited as the only substantive research providing benchmarking energy data for Australian pig and poultry farms. While not cited in Chen et al. (2015), the lamb, wool and grazing beef research provides greater detail than the cited studies and is expected to provide the most recent and comprehensive research covering on-farm energy use in these industries.

With respect to water management, the quantification and hotspot analysis provides similar benchmarking results for Australian livestock as were developed for energy. This research is the first to develop modelled results directly integrating farm dam water balance research with livestock performance to develop meaningful dam extraction rates, providing improved knowledge for assessing dam water supply impacts in Australia.

The water research in this study was also reviewed as part of a recent global assessment of water productivity in livestock systems, which showed this multi-industry dataset to be the most comprehensive LCA research available globally that used farm based measurement and modelling techniques (Drastig et al., in press).

10.6.3 LCA METHOD DEVELOPMENT

With respect to LCA method development, a series of methods applied by the author was subsequently integrated into global livestock LCA guidance, through the author’s participation in the UN FAO LEAP (United Nations Food and Agricultural Organisation, Livestock Environmental Assessment and Performance) program and industry-specific guideline development initiatives (IWTO, 2016; LEAP, 2014; LEAP, 2015a; LEAP, 2015b; LEAP, 2016). While specific contributions are not identified in the guidelines, which are developed by consensus, specific input was given in the development of the following recommendations:
1. Adoption of mass balance methods for predicting manure emissions adopted as best practice in LCA for pork and chicken meat.

2. Recommendation that system boundaries and reference flows are aligned (i.e. inclusion of meat processing and allocation between meat and co-products where the reference unit is a CW product).

3. Recommendation that meat from all slaughter animals, including cull breeding animals, is included in the FU for livestock, because each contributes to the function of providing protein-based food.

4. Recommendation that edible offal, previously omitted from many LCA studies, is included in the reference flow mass, because it contributes to the function of providing protein-based food.

5. Recommendation that all reference flows include specific detail to describe the edible fraction, to avoid errors in comparison between carcase or ‘retail’ products that contain bone.

6. Recommendation that water assessments include supply losses (previously not done by most LCA/water footprint studies in the literature).

7. Provision of benchmarking meat processing data for pigs and poultry.

8. Provision of example water balance calculations for pork production.

10.7 LIMITATIONS, KNOWLEDGE GAPS AND RESEARCH RECOMMENDATIONS

The studies in this thesis represent the first analyses that used comparable methods and covered a range of impact categories and resource uses for the major meat-producing livestock systems in Australia. A range of limitations exist with this research and these limitations are outlined here, with identification of knowledge gaps and future research needs in this area.
10.7.1 **SYSTEM BOUNDARIES**

The study used a system boundary and reference flow that were consistent and reflected a ‘wholesale’ product. Because the system boundary does not extend from cradle-to-grave, burden shifting along the supply chain is a risk, and comparisons should be made with caution (ISO, 2006b). A number of post processing factors limit the comparability of meat products from a consumer perspective. For example, potential differences exist in packaging, transportation, storage, cooking and food waste among the meat products. Additionally, nutritional characteristics differ among meat products and should be considered (Pereira and Vicente, 2013; Williams, 2007). The impact of these factors should be considered in later research if the aim it to inform consumers of the differences between meat types.

10.7.2 **IMPACT CATEGORIES COVERED**

The LCA studies presented here are limited by the narrow assessment of environmental impacts and resources. LCA research aims to be comprehensive in terms of impacts covered (ISO, 2006a; ISO, 2006b), and while the current work covered a wider range than most previous Australian work, gaps remain with respect to impacts on soil health, water quality, biodiversity and factors that directly contribute to negative impacts on human health. Considering these were not included, the risk of unknown trade-offs is high. For example, comparison of industries or industry sectors that rely on grain with those that rely on pasture should ideally include impacts on soil health, water quality and biodiversity because each system has important but different potential impacts. LCA method development is slowly advancing in the areas of soil indicators (Eady et al., 2017; Legaz et al., 2017), biodiversity (Teixeira et al., 2016) and eutrophication (LEAP, 2017; Payen and Ledgard, 2017), but application of these methods in the Australian context will take further development to ensure the methods are meaningful in Australian environments. The following key gaps and research needs exist:

- Eutrophication potential. Research in this area is limited by the availability of meaningful characterisation factors and attenuation factors for Australian
conditions. Ongoing research is taking place to address these gaps by the author (Watson et al., 2018) and others (i.e. Payen & Ledgard, 2017).

- Soil health indicators. Operational soil health indicator methods with manageable data requirements are required that are relevant for Australian conditions.

- Case studies: pending the availability of suitable methods, case studies and regional assessments are required to expand the knowledge base with respect to a wider suite of sustainability indicators for Australian conditions.

- Integration of spatial soil health and biodiversity data with LCA is required to improve the regional representativeness of LCA results.

- Further research is also required to apply new or improved water assessment methods, as new methods become available.

- Toxicity impacts may become important in the future and research is warranted to investigate the methods and data required. This topic is currently a focus of the author’s research (Watson et al., 2018).

In addition to considering a wider range of impact categories, investigation is required to develop a suitable framework for balancing trade-offs between impacts. This could be achieved by developing a normalisation method suited to Australian agricultural production systems, though it is noted that normalisation has no scientific basis (see ISO 14040 and 14044) and would need to be developed with input from a range of relevant stakeholders and decision makers. Nonetheless, this would be valuable for improving the ability of researchers to communicate results to decision makers in Australia, and balance important trade-offs.

10.7.3 LIMITATIONS IN THE CASE STUDY FARM DATASETS

LCA research is data intensive and therefore time consuming both for the researcher and industry participants. The studies focused on major production regions and detailed analysis of a small number of CSF (>80 farms across four species). Limitations with these small datasets were partly addressed, in the case of beef, wool, lamb and pork (chapters 3 - 9) by augmenting the case study results with a
regional or national analysis using larger datasets to mitigate the risk of regional bias in the results.

However, limitations exist because of the small sample size with respect to key factors influencing on-farm water use. For example, the source of water used for pork, lamb and beef production (i.e. dam, bore and river) substantially influenced supply losses and total water use, and varied between farms and regions. Because of the paucity of data regarding water supply in these industries from other studies and the small sample sizes in the present studies, these data are not optimal. More robust results would require collection of water supply data from a much larger sample of farms in all major grazing regions. Considering the cost of this, additional research should investigate developing mathematical relationships between supply losses and easily measured indices to reduce the time and cost of assessing water in LCA research. Likewise, research covering CSF that utilise irrigation for grazing sheep and cattle is required to improve the knowledge base, as these systems contribute significantly to total industry water use.

### 10.7.4 DATA LIMITATIONS

The pig, lamb, wool and beef studies (chapters 3-9) utilised regional or national datasets collected by the Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES) or the Australian Bureau of Statistics (ABS) to improve the representativeness of the results. Similar datasets were not available for chicken meat. Key advantages of applying these datasets include the improved representativeness of these inventory data with respect to herd and flock productivity and purchased inputs, and broader regional coverage. However, limitations also exist with these datasets.

Firstly, in these datasets growth rate and slaughter age of young livestock were not available, and future analyses would be more robust if these data were collected. This could be addressed by adding new categories into the data collection process to determine weight and age of young livestock.
ABARES datasets were utilised to determine purchased inputs and associated impacts for the regional analyses in the wool, beef and lamb studies. These datasets aggregate purchase data into groups (e.g. ‘fodder’ and ‘fertiliser’) rather than specific products and report data on a ‘whole farm’ basis rather than dividing these inputs into sub-systems such as cropping or grazing. The accuracy of disaggregating these impacts and attributing to the relevant sub-system could be improved by adding questions that assist in separating fertiliser and fuel in particular, between major sub-systems (cropping and grazing).

Lastly, these datasets were applied in the present study by averaging data over five years to reduce inter-annual variability and influences on results. While this was appropriate and followed the Australian LCA method, it had the effect of removing the potentially important impact of drought years and high rainfall years. Further investigation of the ‘high and low’ rainfall years is required to provide a more nuanced investigation, and this may prove to be important for understanding the effect of some strategies such as intensification of beef production. It is also noted that these datasets presented herd information as annual inventories that could have masked inter-annual changes in productivity, particularly for beef systems where the life cycle extends over several years. Some researchers (notably, Beauchemin et al. 2010) recommend analysing a beef system over the total life span of the herd (i.e. 8 years in the cited study) to ensure accurate accounting of all impacts from the herd. In the present study, accurate assessment was achieved by ensuring that a steady-state herd or flock was modelled in each instance, which resulted in a similar outcome as achieved by the method applied in the study by Beauchemin et al. (2010).

10.7.5 GEOGRAPHICAL COVERAGE AND PRODUCTION SYSTEM COVERAGE

The studies presented here cover a significant portion of the major production regions for each livestock system, but coverage was variable between the species. Location was found to be sensitive in the assessment of fresh water consumption, water stress, GHG emissions, LU and dLUC. Location may also be sensitive for
water quality indicators and soil health indicators such as salinity, soil acidification, soil erosion and soil carbon (Commonwealth of Australia, 2001), increasing the need for additional, regional assessments for all industries.

With respect to beef production, the current studies have not included the extensive production systems typical in north and north-western QLD, the Northern Territory and northern WA, and case studies in these regions would be beneficial.

Geographical coverage of case study lamb systems was minimal in southern NSW and some parts of Victoria. The effect of this on regionally sensitive impacts such as water was mitigated by applying state-wide datasets from ABARES in the regional analysis, but additional case studies would be beneficial, particularly of irrigated farms. It would also be beneficial to expand the analysis to include national coverage of wool and lamb.

Regarding the pork and chicken meat industry, the largest number of pork CSFs were located in QLD, while no farms were assessed in SA. In order of priority, it would be beneficial to include more case studies from SA, Victoria and southern NSW in future research. Analysis of meat chicken production focused on QLD and SA, and did not include the major NSW, VIC or WA production regions. The effect of this was mitigated by investigating impacts from NSW diets, but further case studies in NSW, Victoria and WA would improve the representativeness of the results.

Considering the significance of location to some impact categories, particularly those related to land (soil and biodiversity) and water (inventory and quality), the integration of spatial data and LCA in the future is a key area of development for LCA databases and agricultural LCA research.

10.7.5.1 Application of Improved Modelling Techniques

LCA is a supply chain modelling technique, rather than a biological modelling technique. Key impacts such as GHG emissions and inputs such as water use were modelled in the present study using accepted, independent methods, but limitations
exist with respect to these models. For example, feed intake and GHG emissions were determined using methods compliant with the National GHG Inventory (Commonwealth of Australia, 2015b) which is known to generate substantially different results for lamb when compared to biophysical models such as GrassGro (see Brock et al., 2013). Similarly, a key aspect of the farm water balance model for predicting runoff to farm dams (specific to ruminant studies) was the K factor method (USDA NRCS, 2007) which does not use a water balance approach for predicting runoff, and has been found to be sensitive, and less accurate than a fully calibrated water balance model for predicting runoff ( Boughton, 1989). However, it is noted that while more detailed hydrological models could be applied to determine water flows, these models also have limitations. Specifically, the number of input parameters are often high, and cumulative model uncertainty is not routinely reported (Benke et al., 2008), potentially leading to an over estimate in the confidence of the results. Research in this area would be advanced through development of consensus models for livestock feed intake and runoff prediction.

Where mitigations are to be modelled, results are more likely to be accurate by applying biophysical models that take into account the changes required to achieve (for example) an increase in liveweight gain in livestock. Integration of biophysical and LCA modelling approaches has been done to a limited extent in Australia (i.e. Brock et al., 2013) and scope exists to expand this, particularly for mitigation research.

10.7.6 Attributional and Consequential Modelling Approaches

Some limitations in the application of this research exist because of limitations in the modelling approach used. All research was completed using attributional LCA, which is suited to benchmarking and hotspot analysis, but is less suited to analyses that recommend significant system change. The state of knowledge for Australian (and global) livestock systems regarding the impacts of interest was minimal when the research was commissioned, and an attributional modelling approach is recommended for providing this primary knowledge base.
However, the LCA research completed in Australia to date is limited when it comes to drawing conclusions that relate to major changes in supply and demand, such as which meat is preferred for meeting future Australian or global demand. Limitations also existed with the type of mitigation practices that could be accurately assessed. For example, carbon sequestration via tree planting in grazing lands to offset livestock emissions may cause a reduction in supply, increasing demand in global markets for beef or lamb. This limitation has been noted, and minimised by avoiding mitigations that were likely to result in major effects on product or co-product supply. However, further research using change-based modelling, or consequential LCA is more suited to examining these impacts.

Future research applying cLCA methods is recommended to explore mitigation of environmental impacts, and to investigate human diet change (from one meat to another, or from meat to vegetable protein). To improve the accuracy of this research, Australian consequential databases are required, and consensus regarding agricultural consequential LCA methods would be highly beneficial to overcome limitations with this method. Recent cLCA studies have been completed, demonstrating this method for Australian pig production (Wiedemann & Watson 2018) and examining the impact of this methodological approach in wool productions systems (Wiedemann et al., 2018c).
11 CONCLUSIONS

Food production is an imperative for humanity and is known to generate environmental impacts. In contrast, other activities related to comfort or leisure, such as an overseas flight, generate environmental impacts without being vital.

Meat production and consumption contributes substantially to global food supply, and also to environmental impacts and resource use. Meat consumption is important not only for meeting nutritional needs but also for cultural reasons.

Reducing environmental impacts is an important priority for meat production systems. Doing so requires a sound understanding of the impacts generated by current production systems from region to region. The results presented in this thesis reported environmental impacts, hotspots and potential mitigations across the four major meat production systems, substantially contributing to this knowledge base.

This research has shown that simple trends do not always exist among different meat production systems. While results showed that GHG emissions were higher for ruminants than for monogastrics, this inter-species difference was less distinct for energy demand and water stress. This showed that GHG was not a suitable proxy indicator for the wider suite of indicators assessed. Crop land occupation was higher for monogastrics, and HEPR was also higher for monogastrics compared to grass-fed ruminants. This contrast between GHG emissions, HEPR and crop land, with followed divergent trends between the meat production systems, was the most significant trade-off observed in this research. Because monogastrics rely on grain from crop land, it is not possible to produce these meats from rangeland areas, which are extensive in Australia, indicating that the best use of these areas, from a food production point of view, remains to graze ruminants.

This said, the problem of high GHG emissions from ruminants remains a significant concern that must be addressed by the red meat and wool industries. LCA research is well suited to providing full system context for GHG research, and also for identifying potential trade-offs. Consequently, research groups are encouraged to develop and integrate “life cycle thinking” and LCA practice into mitigation research.
teams in the future. In this respect LCA has two important and different roles that have been exemplified for the industries studied in this thesis: firstly, LCA can be used to lead research by defining sensitive and uncertain emissions and directing research that can verify these emissions and also move towards mitigation. Secondly LCA can be applied to summarise knowledge into a cohesive whole, which has again been demonstrated in this thesis for the industries in focus.

This research found that environmental impacts were regionally variable, and consequently research must be regionally specific to develop a rounded understanding of environmental sustainability. Applying research from one part of the country to another part may result in different trade-offs than existed in the initial context. At the global scale this challenge is multiplied exponentially, indicating that regional research is needed before generalisations are made about one meat type or another.

The research presented here has shown that water use from ruminants is only modestly higher than monogastrics, and in some regions, water stress impacts are similar among all meat production systems. Indeed, regional impacts were more significant than the production system or meat type in some cases. These results indicate that regional or global generalisations that beef systems use large amounts of water compared to other meat types are not strongly supported in the Australian context.

The research has identified that a wider range of impact categories need to be assessed in LCA research to provide a more robust assessment of impacts. This will create more complexity and more trade-offs, which can be addressed by prioritising between impact categories or using a normalisation approach to help decision making. This would be advanced if an agreed set of normalisation factors were developed for regional Australia. It is noted that priorities will also vary from region to region. For ruminant production in rangeland areas, where few alternative uses exist for water and crop land, these factors will be less significant than GHG emissions, which tend to be high in these regions. Thus, utilising isolated irrigation water supplies and small areas of arable land to improve livestock productivity and reduce GHG emissions could be a sustainable solution. However, in regions where
grain and forage crops are grown using irrigation from stressed water sources, water will be a higher priority and potential trade-offs between water stress and GHG emissions need to be considered.

This research has identified hotspots and mitigations that can be applied in Australian production systems to reduce environmental impacts. When a range of trade-offs were considered, some mitigation strategies were found that reduced multiple impacts concurrently, and these are described separately for monogastrics and ruminant systems in the following sections. Research and extension should focus on these strategies that have multiple benefits, rather than one impact (such as GHG emissions) only.

**Pork and Chicken Meat Production**

Australian pork production is a modern, multi-faceted industry with different production systems suited to different regions and market preferences. As with all livestock industries, it produces high quality food, predominantly for Australia’s domestic population rather than export. This industry, more than most others, was found to have opportunities to reduce impacts by changing production systems to known alternatives. Clearly biogas with energy generation is a breakthrough technology, reducing GHG emissions and energy demand substantially. Opportunities exist to leverage this technology to ‘close the loop’ in other respects by reducing on-site water use via recycling and improving nutrient cycling. The pig industry also has an important efficiency role in food production because it can utilise food waste, turning this into a valuable meat product. This opportunity could be dramatically expanded. For example, more than half of the industry’s feed needs could be met via waste food if very efficient systems were developed, potentially integrating insects or other waste processing methods.

The deep litter production system for pigs showed that lower production intensity could reduce environmental impacts. There are challenges for producers in managing pigs on deep litter and addressing these challenges via future research could increase adoption and therefore reduce environmental impacts. This should be considered by industry, though it is noted that conventional housing with biogas generates better outcomes and should be adopted in preference to deep litter housing. Outdoor pig
production showed potentially lower GHG emissions impacts than conventional production (without biogas) but the risk of trade-offs is noted, and ongoing research efforts are needed to understand and manage these into the future.

With respect to production efficiency, opportunities also exist for reducing FCR in pork production, which will, all things being equal, lead to lower environmental impacts throughout the supply chain. This is a high priority for the industry from the perspective of reducing production costs and should also be a high priority from the perspective of sustainability. More research attention is needed on integrating “production” research, such as nutrition, with environment research, to ensure gains occur in both sectors. This is another instance where integrating LCA research into broader research programs is recommended.

The Australian chicken meat industry is modern and well developed, with world class production efficiency and FCR, leading to relatively low environmental impacts. Opportunities exist for this industry to leverage the knowledge developed in the pork industry to reduce these further. For example, utilising energy in manure will reduce GHG emissions and energy demand and is an area where research should be directed. While the industry has traditionally been located near capital cities, this is changing, and regulatory constraints around on-site manure handling in regional areas may diminish as the industry moves production centres to regional areas. This opens the possibility for manure energy generation, and biogas is a mature option to investigate and apply in the industry today.

Understanding environmental impacts associated with diet is important for the chicken meat industry. Research presented and discussed in this thesis has shown that some diet components (such as imported soymeal) contribute a disproportionately high amount of the environmental impacts from chicken. Allied research has also shown that improvements in diet formulation by reducing crude protein can reduce these. As with the pork industry, this highlights the need for environmental considerations to be seriously considered when formulating rations and conducting nutrition research. A major advance could be made in this area via a joint research effort from the meat chicken, pork and egg industries to develop a rigorous understanding of sustainable diets for Australian production.
**Beef and Lamb Production**

Australian beef production is a major agricultural industry producing high quality protein food for Australia and the world. It is a major industry with production in all states and a large variety of climatic zones. The strength of ruminant grazing animals is the production of high quality protein food from low quality forage that is not edible for humans or monogastric livestock species, produced on rangeland areas not suitable for growing crops. The concomitant weakness of ruminant digestion is production of methane, which has proven difficult to mitigate. Clearly research is needed to mitigate enteric methane, ideally via options that increase resource use or other impacts minimally, and LCA research can help inform this research. However, other options such as increasing carbon sequestration are also important. Soil carbon has been shown to be a potentially significant mitigation in southern Australia (investigated in lamb production in the present studies but relevant for beef also) and this should not be discounted though it is difficult to achieve in practice.

A fundamental trade-off presently exists for ruminants in general between utilising crop land (or grain) and GHG emissions. The Australian industry has intensified production over time, leading to lower GHG emissions impacts but higher impacts in other areas. This has been assisted by adopting grain finishing, which helps to improve production consistency and has benefits for product quality. Grain finishing and intensive grazing, particularly in mixed cropping regions, has achieved this largely by utilising energy and crop land, and to some extent irrigation water resources. Clearly grain finishing can improve production efficiency, but the trade-offs need to be considered and mitigated also, and at the micro scale it could be argued that other meat products are more efficient at utilising grain. However, the ‘big picture’ has not been painted with respect to intensification in the beef industry yet, as alluded to in this thesis. For example, utilising less land may enable the industry to reduce deforestation and to reverse this by sequestering carbon, reducing industry wide impacts.

Another potentially important role for grain finishing is in droughts, when the benefits (in terms of GHG emissions, and possibly energy, land impacts and water) may be much higher than suggested by the detailed system comparisons presented here, which were averaged over a number of years and large regions, reducing the
effect of low rainfall years. Clearly, the basic system knowledge and framework has now been developed, but the transformative questions have not been asked in the LCA research space and ongoing research is needed (and underway) to investigate these transformative opportunities. These big questions also require new analysis methods such as cLCA to be adopted by industry to explore and avoid the potential for trade-offs.

However, future investigations must keep in focus the key benefit of beef: utilising low quality forage from rangelands not suitable for crops. Thus, intensification is best done by leveraging this strength, utilising the smallest amount of grain or crop land possible to maximise herd efficiency. The HEPR indicator is useful to help industry keep this trade-off in mind, though more work is needed to define the amount of grain and indeed crop land that can be utilised beneficially by livestock, such as in years when droughts during the growing season or rainfall during harvest lead to crop failure or downgraded grain which is not suitable for human consumption but is well suited as a feed for livestock is produced.

**Summary**

At a high level, this research identified a series of mitigation options for each meat production system that reduced impacts across multiple impact categories. Research and adoption strategies should focus on these opportunities, and cLCA methods should be applied to improve understanding of indirect effects. The research showed that water use and water impacts associated with meat production systems was lower in Australia than suggested by the global livestock water footprint literature. Specific concerns relate to the use of highly stressed irrigation sources for feed grain and fodder production in southern Australia, and further investigations into these regions and production systems are warranted.

Across all industries, a broader understanding of environmental impacts associated with meat production is vitally needed to avoid trade-offs and to identify the most suitable mitigation strategies, considering multiple impacts. Prioritisation between different environmental impacts is also needed, and this requires input from policy makers.
While this research focused on meat production systems through to production of a wholesale product, further research is required to cover the full ‘cradle-to-grave’ to inform consumers of the impact of meat consumption. Where information is sought to inform a choice between one meat and another, new analyses using eLCA methods are required. To advance this, there is an urgent need for new methods, data and research capacity to be developed in this area.
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Analysis of resources and GHG emissions from meat production


Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


Wiedemann, S., McGahan, E. J. and Poad, G. (2012b) *Using Life Cycle Assessment to Quantify the Environmental Impact of Chicken Meat Production*, Australia: Rural Industries Research and Development Corporation (RIRDC).


APPENDIX 1

12.1 CITATIONS

The published papers included in this thesis have been cited by various other literature since publication. Reporting this was a university requirement for enrolment in a thesis by prior publication. At the time of writing, the publications contained in this thesis had been released for a maximum of 2.2 years, and most had been available for 1-1.5 years in the “on-line early” format. The number of citations for each publication (as of the 20-03-2018) was collated from data analytics available through Google Scholar, and provided in Table 9 of Appendix 1.

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<tr>
<th>Publication</th>
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<td>2015</td>
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<td>Resource Use and Environmental Impacts from Beef Production in Eastern Australia Investigated Using Life Cycle Assessment</td>
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<tr>
<td>Resource Use and Environmental Impacts from Australian Export Lamb Production: A Life Cycle Assessment</td>
<td>2016</td>
<td>13</td>
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<tr>
<td>Resource Use and Greenhouse Gas Emissions from Three Wool Production Regions in Australia</td>
<td>2016</td>
<td>3</td>
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<tr>
<td>Resource Use and Environmental Impacts from Australian Chicken Meat Production</td>
<td>2017</td>
<td>10</td>
</tr>
<tr>
<td>Environmental Impacts and Resource Use from Australian Pork Production Determined Using Life Cycle Assessment: GHG Emissions</td>
<td>2016</td>
<td>6</td>
</tr>
<tr>
<td>Environmental Impacts and Resource Use from Australian Pork Production Determined Using Life Cycle Assessment: Energy, Water, Land Occupation</td>
<td>2017*</td>
<td>Nil (only currently published online)</td>
</tr>
<tr>
<td>Resource Use and Greenhouse Gas Emissions from Grain Finishing Beef Cattle in Seven Australian Feedlots: A Life Cycle Assessment</td>
<td>2017*</td>
<td>Nil (only currently published online)</td>
</tr>
</tbody>
</table>

*a Citations based on reports of Google Scholar, 20-03-2018.

*This date refers to the year the publication was made available on the publishing journal’s website, but they are yet to be included in the printed journal.
12.2 ATTRIBUTION STATEMENTS
Attribution statement

The paper Environmental impacts and resource use of Australian beef and lamb exported to the USA determined using life cycle assessment was led by S.G Wiedemann, and co-authored by E.J. McGahan, C.M. Murphy, M.J. Yan, B.K. Henry, G. Thoma & S. Ledgard.. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions, water and land
- Data analysis and modelling of meat processing, including development and application of allocation rules
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McGahan contributed to the study and manuscript via:

- Review of the goal and scope phase
- Review of the manuscript

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- Assistance with graphs, tables and manuscript preparation.

Yan contributed to the study and manuscript via:

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Ledgard contributed to the study and manuscript via:

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Thoma contributed to the study and manuscript via:

- Review of the goal and scope phase
- Provision of data and unit processes pertaining to shipping and road transport in the USA, and cold storage in the USA.
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Approved by:  CM Murphy

Signed:  Date: 31/05/2016
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Signed: [Signature] Date: 29/15/2016
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Approved by: G Thoma

Signed: G Thoma  Date: 28 May 2016
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- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and herd outputs
- primary data acquisition and data analysis
- preparation of the manuscript and completion of the peer-review process

McGahan contributed to the study and manuscript via:

- Valuable assistance to development of the on-farm water modelling methodology
- Assistance with data collection
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation.

Yan contributed to the study and the manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation.

Approved by: C. M. Murphy

Signed: [Signature]  Date: 31/05/2016.
The paper Resource Use and Environmental Impacts from Australian Chicken Meat Production was led by S.G Wiedemann and co-authored by E.J McGahan and C M Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
- Development of the manure management system model
- Development and application of grain processes used in the modelling of feed inputs
- Data acquisition and data analysis, including impact assessment
- Modelling of grow out units and meat processing
- Allocation methodology development
- Preparation of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model

Murphy contributed to the study and manuscript via:

- Assistance with graphs and manuscript preparation

Approved by: E.J McGahan

Signed: [Signature] Date: 29/5/2016
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- Assistance with manuscript preparation.

Yan contributed to the study and the manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation.

Approved by: M.J. Yan

Signed: ____________________________ Date: 30 May 2016
Attribution statement

The paper Resource use and greenhouse gas emissions from grain-finishing beef cattle in seven Australian feedlots: a life cycle assessment was led by S.G Wiedemann, and co-authored by R. Davis, E.J. McGahan, C.M. Murphy and M Redding. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions
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- Integration of primary data into the LCA model, and data analysis
- Preparation of the manuscript and completion of the peer-review process

Davis contributed to the study via:

- Assistance with integrating primary data from previous inventory collection projects into the LCA analysis.
- Review of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data interpretation
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with data analysis.
- Assistance with manuscript preparation.

Redding contributed to the study and the manuscript via:

- Assistance with integration of manure emission factors into the modelling.
- Review of the manuscript.

Approved by: **Rodney James Davis**

Signed: **R.J. Davis**  Date: 20/03/2018
Attribution statement

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Approved by: Caolínn Murphy

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Date: 20/03/2018
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Approved by: [Signature]

Signed: [Signature] Date: 22/3/2018
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Approved by:  

Eugene McGahan

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EJM McGahan  

Date: 21/3/2018
Attribution statement

The paper *Resource use and environmental impacts from Australian export lamb production: a life cycle assessment* was led by S.G Wiedemann, and co-authored by M-J Yan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

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- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and flock outputs, and application of novel allocation methods to handle co-production of wool and live weight.
- primary data acquisition and data analysis
- preparation of the manuscript and completion of the peer-review process

Yan contributed to the study and manuscript via:

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- Assistance with development of figures and tables for the manuscript

Murphy contributed to the study and manuscript via:

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- Assistance with manuscript preparation.

Approved by: C. M. Murphy

Signed: [Signature]
Date: 31/05/2016.
The paper *Environmental impacts and resource use from Australian pork production assessed using life cycle assessment: 1. Greenhouse gas emissions* was led by S.G Wiedemann, and co-authored by E.J McGahan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
- Development and application of the greenhouse gas model
- Development of the water use model
- Development and application of grain processes used in the modelling of feed inputs
- Data acquisition and data analysis, including impact assessment
- Allocation and uncertainty analysis
- Preparation of the manuscript

McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model and herd modelling

Murphy contributed to the study and manuscript via:

- Assistance with data analysis and assistance with preparation of figures, tables and manuscript.

Approved by: E.J McGahan

Signed: [Signature] Date: 29/5/2016.
Attribution statement

The paper Resource use and environmental impacts from beef production in eastern Australia investigated using life cycle assessment was led by S.G Wiedemann, and co-authored by E.J. McGahan, C.M. Murphy and M.J. Yan. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling herds and impacts from greenhouse gas emissions
- Development and application of novel methods for determining water use from farm storages
- Development of novel methods for the determination and categorisation of land use
- Development of methods for sub-dividing farm systems and herd outputs
- primary data acquisition and data analysis
- preparation of the manuscript and completion of the peer-review process

McGahan contributed to the study and manuscript via:

- Valuable assistance to development of the on-farm water modelling methodology
- Assistance with data collection
- Review of the manuscript

Murphy contributed to the study and manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation.

Yan contributed to the study and the manuscript via:

- Assistance with data analysis
- Assistance with manuscript preparation.

Approved by: C. M. Murphy

Signed: ___________________________ Date: 31/05/2016.
The paper *Environmental impacts and resource use from Australian pork production assessed using life cycle assessment: 1. Greenhouse gas emissions* was led by S.G Wiedemann, and co-authored by E.J McGahan and C.M. Murphy. In this collaborative publication, Wiedemann was responsible for the following aspects:

- Development of the goal, scope and methodology for modelling the production system
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McGahan contributed to the study and manuscript via:

- Assistance with data collection and analysis
- Assistance with application of the manure excretion model and herd modelling

Murphy contributed to the study and manuscript via:

- Assistance with data analysis and assistance with preparation of figures, tables and manuscript.

Approved by: E.J McGahan

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Co-author statement regarding publications submitted for a thesis

As a co-author to the following paper(s) I certify that I agree to the paper(s) being submitted as part of a thesis by S.G. Wiedemann.

Name: Rooney James Davis

Signature: [Signature]

Date: 20/03/2018


Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


Co-author statement regarding publications submitted for a thesis

As a co-author to the following paper(s) I certify that I agree to the paper(s) being submitted as part of a thesis by S.G. Wiedemann.

Name: Beverly K. Henry

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Date: 23 March 2018


Co-author statement regarding publications submitted for a thesis

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Name: Stewart Ledgard

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Date: 20 March 2018


Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


Co-author statement regarding publications submitted for a thesis

As a co-author to the following paper(s) I certify that I agree to the paper(s) being submitted as part of a thesis by S.G. Wiedemann.

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Date: 21/3/2018


As a co-author to the following paper(s) I certify that I agree to the paper(s) being submitted as part of a thesis by S.G. Wiedemann.

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Date: 20/03/2018


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Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


Co-author statement regarding publications submitted for a thesis

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Date: March 19, 2018


Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


Co-author statement regarding publications submitted for a thesis

As a co-author to the following paper(s) I certify that I agree to the paper(s) being submitted as part of a thesis by S.G. Wiedemann.

Name: Mingjia Yan

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Date: 20/3/15


Wiedemann, S., McGahan, E. and Murphy, C. (2017b) 'Resource Use and Environmental Impacts from Australian Chicken Meat Production', *Journal of Cleaner Production*, 140, pp. 675-684.


APPENDIX 2

BEEF CATTLE MITIGATION SCENARIOS

The following section provides background data used to develop the mitigation scenarios presented for beef cattle in section 10.5.

A series of scenarios were investigated to explore options to reduce GHG emissions from backgrounding and finishing. These scenarios were based on approaches that could be applied in Queensland, though not specifically at the properties where the supply chains were located. Production data and assumptions are provided in Table 10.

Table 10. Livestock production characteristics for 7 scenarios for finishing export steers

<table>
<thead>
<tr>
<th>Finishing system</th>
<th>Growth rate</th>
<th>Age at slaughter (months)</th>
<th>Slaughter weight (kg LW)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Backgrounder phase – kg/d</td>
<td>Finishing phase – kg/d</td>
<td>Birth to slaughter – kg/d</td>
</tr>
<tr>
<td>Forage/suppl. and feedlot finishing (75d)</td>
<td>1.2</td>
<td>1.7</td>
<td>0.83</td>
</tr>
<tr>
<td>Forage/suppl. and feedlot finishing (120d)</td>
<td>1.2</td>
<td>1.70</td>
<td>0.88</td>
</tr>
<tr>
<td>Leucaena, feedlot finishing</td>
<td>1.1</td>
<td>1.7</td>
<td>1.10</td>
</tr>
<tr>
<td>Grass, Forage and supplementary feed</td>
<td>0.8</td>
<td>0.9</td>
<td>0.71</td>
</tr>
<tr>
<td>Leucaena with revegetation</td>
<td>0.75</td>
<td>0.75</td>
<td>0.68</td>
</tr>
<tr>
<td>Irrigated leucaena and supplementary feed</td>
<td>1.1</td>
<td>1.1</td>
<td>0.79</td>
</tr>
<tr>
<td>Low input grass finishing</td>
<td>0.455</td>
<td>0.35</td>
<td>0.47</td>
</tr>
</tbody>
</table>

For the feedlot scenarios, the diets were formulated to achieve oil inclusion rates of 6% of DMI, as an enteric methane mitigation strategy. The reduction in enteric methane was determined using the relationship reported by Beauchemin et al. (2008).

For the paddock supplementation scenarios, lipids were also included to mitigate
enteric methane by feeding a cotton seed and grain ration. Dietary oil levels were 2-3%.

Leucaena scenarios were modelled based on data reported by Radrizzani et al. (2011a; 2011b) and Shelton and Dalzell (2007). The leucaena scenarios assumed 11% lower enteric methane after Kennedy and Charmley (2012). The scenario including regrowth assumed that the higher stocking density achievable with leucaena left some land available which could be set aside and allowed to regrow with minimal inputs. For each hectare of leucaena, an assumed ½ ha. of land was available for regrowth. Regrowth was assumed to be Acacia (Brigalow) open forest, with carbon storage of 32 t per hectare (after Fensham and Guymer, 2009). Carbon sequestration was annualised over 100 years to provide an annual rate of 0.32 t C ha. yr.

Further detail can be found in Wiedemann et al. (2013).