Final report

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Prepared by:	SG Wiedemann, CM Murphy, EJ McGahan, M Renouf, P Prasad, SL Bonner, EN Zadow and B Henry.
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Meat & Livestock Australia Limited
Locked Bag 991
NORTH SYDNEY NSW 2059

Northern Australian Beef Supply Chain Life Cycle Assessment – Final Report

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Abstract

This study completed a Life cycle assessment (LCA) investigating resource use and environmental impacts from beef production in two Queensland supply chains, from production on-farm through to consumption either in Australia or Japan. The study investigated energy demand, water use, land occupation, eutrophication potential, soil depletion potential and greenhouse gas emissions. We divided the supply chain into three sections and presented results for each: the first being 'production of live weight beef at the farm gate', the second being 'production of boned beef ready for wholesale/retail' and the last being 'beef consumed in the home, either in Australia or Japan'.

At the farm gate, energy, water and GHG were similar to previous studies of Australian beef and international studies. Energy demand as primarily associated with purchased inputs (i.e. feed supplements and services) and farm energy use (i.e. diesel and electricity use). Water use was primarily associated with direct drinking water requirements for livestock, and storage losses (evaporation) from farm dams. Land occupation was divided into arable and non-arable land resources. Of these, the arable land occupation was minimal, though rangeland occupation was substantial. Arable land resources are constrained in Australia and globally, and this is a critical resource for food production. The sustainable and efficient use of these land resources is critical for maintaining global food production. Results show comparable GHG emissions to other recent Australian studies at the farm gate (11.2-12.9 kg CO_2-e / kg LW). While nutrient loss to waterways is a topic of national concern in Australia, there was insufficient primary research available to develop characterisation factors and quantify eutrophication for the supply chains investigated in this study. Therefore, Eutrophication Potential was qualitatively assessed for each farm. The authors determined that the eutrophication potential from beef production was very low for these farms because of the very low stocking rates and negligible fertiliser inputs.

Throughout the supply chain, the absolute value of the impacts increased with a change of functional unit e.g. from live-weight to boned beef, or from boned beef to beef consumed at the home.

Executive Summary

Life cycle assessment (LCA) is a powerful tool for investigating system efficiency and identifying the environmental impacts associated with a product such as beef. This project extends LCA research for Australian red meat to two Queensland beef supply chains, from production through to consumption. Results are presented using a number of mid-point indicators representing different stages of the supply chain (per kilogram of live-weight at the farm gate, per kilogram of bone beef at the retail shelf, and per kilogram of beef consumed in the home). The study covered the following resource use and environmental impact indicators: Cumulative Energy Demand, Consumptive Water use, Stress Weighted Water use, Land occupation, grain use/use of human edible energy and protein, Eutrophication Potential, Soil Carbon Flux Potential, Soil Depletion Potential and total GHG emissions. This is the most comprehensive study of its type for northern beef production to date. The two supply chains produced different products, with the north east (NE) supply chain producing grass fed bullocks (602kg LW at slaughter) for the Japan ox market, while the south west (SW) supply chain produced grain finished beef (434 kg LW at slaughter) for the domestic market.

Results – Farm Gate (per kilogram of live weight – LW)

Energy demand ranged from 4.3 ± 5% to 4.7 ± 8% MJ / kg LW. Consumptive water use ranged from 183-248 ± 35% L / kg LW. This assessment of water use included drinking water requirements and water supply losses (evaporation from dams), together with water use associated with the production of inputs such as electricity. Stress weighted water use was considerably lower than consumptive water use, ranging from 7.7-45.9 L H_2O-e / kg LW. Stress weighted water use is a measure of the impact of using water. Where pressure on water resources was considerably lower than the global average, the apparent water use is considered to be lower. Consumptive water use and stress weighted water use assessed using LCA generated results that were orders of magnitude lower than most estimates of 'virtual water' or the water footprint for beef cattle. The main difference in these methodologies was the handling of rainwater used to grow crops and pastures (so called 'green' water associated with water loss by evapotranspiration), which is included in a virtual water / water footprint assessment but is not considered a source for estimating consumptive water use in LCA, or in the general understanding of water use used in society.

Land resource use was assessed, dividing land into arable and non-arable land occupation. Data have not been reported by other researchers using these categories. Arable land occupation ranged from 0.5-3.9 m^2 / kg LW for the NE and SW QLD farms respectively, while non-arable land occupation was considerably higher because of the low stocking densities used on each farm.

Total land occupation (the combination of arable and non-arable land use), was higher than values reported in the literature for European beef production, though we consider this measure to be of less relevance for assessing the use of scarce land resources or impacts on biodiversity, because of the considerable differences in management and impacts from grazing on largely unmodified rangelands compared to cultivation. Further work is required to understand the impact of land occupation in the Australian agricultural industries.

Greenhouse gas emissions ranged from 11.2-12.9 kg CO_2-e / kg LW, with the lower emissions coming from the SW supply chain which utilised grain finishing and had higher levels of herd productivity (weaning rates and growth rate to slaughter). A number of GHG mitigation strategies were investigated, providing reductions of up to 31% in GHG emissions. Where sequestration potential was included, the mitigation potential was higher. Most sequestration scenarios relied on utilising other resources such as energy, arable land, grain or water to achieve productivity

improvements and subsequent reductions in GHG. This showed a trade-off in objectives and also highlights that some mitigation strategies will be limited by the resources available.

The farm gate results were broadly similar to previous Australian beef LCA research for GHG emissions intensity and water use. Compared to the international literature, our results were lower in energy use and similar to lower in GHG emissions intensity. Consumptive and stress weighted water use have not been studied extensively in the international literature and comparisons could not be made.

Results – Retail Shelf (per kilogram of boned beef)

Results at the retail shelf take into account for meat processing and transport, storage and wastage associated with retail distribution. The primary difference compared to results 'per kilogram of live weight' is associated with the loss of product mass at the point of meat processing and processes used to account for co-products.

Energy demand ranged from 15.2 ± 6% to 16.8 ± 4% MJ / kg boned beef. Consumptive water use ranged from 356 ± 38% to 496 ± 37% L / kg boned beef, while stress weighted water use ranged from -4.5 to 77 L H_2O-e / kg boned beef. Occupation of arable land ranged from -1.9 ± 12% to 4.9 ± 14% m^2 / kg boned beef. Human edible energy (-8.3 to 10.4 MJ / MJ boned beef) and protein (-0.1-0.05 kg / kg boned beef) provided an indication of the requirements for human edible inputs per kilogram of meat produced. The negative values for stress weighted water use, arable land use and human edible energy and protein require further explanation. The negative values resulted from the system expansion approach used to handle co-products. This approach 'expands' the production system to account for avoided products (namely soybean meal and canola oil) that substitute for meat processing co-products of meat/blood/bone meal and tallow. Where beef co-products were substituted for soymeal grown in Australia and the USA, with a proportion of irrigation water from stressed catchments, the offset for stress weighted water use was high. Similarly, the offset for arable land use, human edible energy and human edible protein were also high. This highlights that co-products from beef production play an important role in the Australian vegetable protein meal market by reducing demand for soymeal, much of which is imported. Where negative values were reported, this effectively offset all the impacts of producing beef. Results are also presented using alternative (mass and economic) methods for handling co-products, which result in different impacts.

Greenhouse gas emissions ranged from 11.2-12.9 kg CO_2-e / kg LW, with the lower emissions coming from the SW supply chain which utilised grain finishing and had higher levels of herd productivity (weaning rates and growth rate to slaughter). A number of GHG mitigation strategies were investigated, providing reductions of up to 31% in GHG emissions. Where sequestration potential was included, the mitigation potential was higher. Most sequestration scenarios relied on utilising other resources such as energy, arable land, grain or water to achieve productivity improvements and subsequent reductions in GHG. This showed a trade-off in objectives and also highlights that some mitigation strategies will be limited by the resources available.

Following assessment of meat processing, the magnitude of the impacts increased with a change of functional unit. Energy demand was high from meat processing, while most other impacts were relatively low.

We investigated the consumption of human edible protein and energy to produce beef, as a measure of the net food production from the beef sector. Ruminant livestock fed entirely on grain will have a relatively poor conversion efficiency for feed inputs to outputs. However, the cattle investigated in this study (even those finished with grain) consume very little of their total feed requirements as grain. The result is that the net production of human edible protein was considerably higher than the amount consumed throughout the system, demonstrating a net contribution to food production. Interestingly, because co-products from meat processing (such as tallow and meat meal) displace human edible energy and protein products, the grass finished cattle were found to generate more human edible protein not only via the primary meat product, but also via the displacement of soy and canola by co-products.

Through to the point of consumption of beef in the home (in Australia or Japan) impacts were greatest from primary production phase for all impacts with the exception of energy demand. The largest impacts post processing were from food wastage (at retail and in the home). Transport to Japan was found to generate only minor contributions to GHG (<2%).

This study applied a novel, hybrid approach to handle co-products. Hides and human edible meat (both carcase and human edible offal) were handled using mass allocation. This process applies the same burden to products based on product mass. A ‘system expansion’ approach was applied to handle minor co-products

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List of Abbreviations

ABS – Australian Bureau of Statistics

CH_4 – Methane

CO_2 – Carbon Dioxide

DCCEE – Department of Climate Change and Energy Efficiency

EP – Eutrophication Potential

GHG – Greenhouse Gas

GWP – Global Warming Potential

HSCW – Hot Standard Carcass Weight

IPCC – Intergovernmental Panel on Climate Change

LCA – Life Cycle Assessment

LCI – Life Cycle Inventory

LPG – Liquid Petroleum Gas

MLA – Meat & Livestock Australia

N_2O – Nitrous Oxide

NGGI – National Greenhouse Gas Inventory

VW – Virtual Water

WF – Water Footprint

1 Introduction

1.1 Background

Meat and Livestock Australia Ltd (MLA) have commissioned many projects investigating environmental issues, using Life Cycle Assessment (LCA) and other research approaches. These projects have been commissioned to enable the industry to quantify and improve environmental performance and provide credible information to the industries' supporters and critics. The industry also realises that in the future, both domestic and international customers may demand information on the environmental credentials of Australian beef, and it is the responsibility of the industry to provide this information.

While a considerable amount of research is undertaken in these areas, few projects are able to provide an overview of a number of environmental issues at the same time, covering the whole supply chain. For a complex, dynamic system such as a beef supply chain, it can be difficult to understand how changes in one practice may influence others. This is particularly relevant for research areas that bridge multiple research fields. LCA is a useful tool for drawing these research areas together, quantifying impact areas and mitigation potential, and providing results in the context of beef production.

This project follows on from several projects commissioned by MLA and conducted by FSA Consulting as the lead or associate research agency. These provide important background to this project and are the source of some methods and data. Rather than reproducing this work, an outline of key projects and reports is supplied in this section and will be referred to where relevant in the report.

COMP.094 – Life Cycle Analysis of the Red Meat Industry – Commissioned in late 2004 and completed in 2009 (led by UNSW with FSA Consulting as a project team member).

This project covered three southern supply chains. These were: A Victorian organic beef operation (high rainfall, high production), a southern NSW beef operation with steers finished on grass or in the feedlot (moderate rainfall, moderate-high production) and a Western Australian lamb operation (low rainfall, some grain finishing). Full details from this study can be found in the original reference (as cited byPeters et al. 2009) or from the peer reviewed journal articles covering greenhouse gases / carbon footprint (Peters et al. 2010a) and water use (Peters et al. 2010b).

B.CCH.2022 – Review of Water Use and GHG Emissions from Red Meat Production – Commissioned February 2009 and completed August 2009 (led by FSA Consulting).

This project was presented as three reports:

Report 1 – GHG and Water Usage Review. This report provides an overview of the topic from an industry wide perspective, using an extensive literature review of assessment frameworks, policy and supply chain level reporting in the literature (i.e. life cycle assessment). This report also contains technical reviews of energy usage, the processing sector and vegetation management.

Some information from this report has been summarised here, and further information can be found from the original reference (see Wiedemann et al. 2010a).

Report 2 – Enteric Methane Review. This report focused on enteric methane alone because of the significance of this emission source to the red meat industries. The report was compiled by Dr David Cottle and Professor John Nolan from the University of New England (UNE), covering nutritional and genetic approaches to the mitigation of emissions from livestock, modelling of livestock emissions and a review of the Department of Climate Change methodologies available for the red meat industries. Further information can be found from the original reference (Cottle & Nolan 2009).

Report 3 – Nitrous Oxide and Carbon Cycling in Soils and Waste Review. This report was compiled by Dr Matt Redding, and covers all emissions related to nitrous oxide and (non-enteric) carbon emissions from across the red meat supply chain, with particular attention to the feedlot sector. Further information can be found from the original reference (Redding 2009).

B.FLT.0339 – Water and energy usage for individual activities within Australian feedlots (FSA Consulting).

This project conducted an in-depth assessment of water and energy use at Australian feedlots, including collection of production data over a 2 year period. These data provide some input data for the rapid assessment in this report. Further information can be found from the original reference (Davis et al. 2008a, b).

B.FLT.0360 - A Scoping Life Cycle Assessment of the Australian Lot Feeding Sector (FSA Consulting).

This project focused on the feedlot sector of the supply chain, investigating water, energy and Global Warming Potential (GWP) with particular reference to feedlot manure management. This, along with other feedlot specific research, will be utilised in this project to strengthen the feedlot comparison with grass-fed beef. More detailed findings are available in the MLA publication (Wiedemann et al. 2010c).

1.2 Project Objectives and Reporting

The project has the following broad objectives:

- To quantify the environmental impacts of two Queensland beef supply chains for domestic and premium export beef.
- To produce credible data on climate change impacts and water use in these supply chains (to the general public).
- To enable a comparison between grain and grass finishing supply chains.
- To identify key environmental risks and quantify the likely gains by applying GHG mitigation strategies currently available to the industry – such as changes to nutrition.
- To cover other relevant environmental and resource issues such as energy usage, land use, soil erosion and eutrophication.

This final report covers all objectives of the project.

2 Life Cycle Assessment

Life cycle assessment is a multi-criteria, whole supply chain analysis tool used for assessing the resource use and environmental impacts associated with producing, using and disposing of a product or a service. LCA was developed for use in the manufacturing and processing industries, and was applied to food production systems (and therefore agriculture) more recently. There has been a rapid increase in the number of agriculture and food related LCA studies over the past 10 years. Life cycle assessment is a well-established research method, defined by a number of international and Australian standards. However, the broad objectives and comparatively recent application to food production mean that methodology development is on-going.

The applications of LCA research are broad, ranging from comparison of the environmental credentials of a product through to system auditing and directing research. LCA can be used as a theoretical approach to compare mitigation scenarios for research or for comparing materials during the evaluation of a new product. The 'whole life cycle' focus allows LCA to identify (and help avoid) 'burden shifting' between either: i) different stages in the supply chain, ii) different environmental impacts, or iii) between different geographical locations or industries.

2.1.1 LCA Research Framework

International standards have been developed to specify the general framework, principles and requirements for conducting and reporting LCA studies (ISO 2006a: 14040) and (ISO 2006b: 14044). The framework includes four aspects:

- Goal and scope definition: The product(s) to be assessed are defined, a functional basis for comparison is chosen and the required level of detail is defined.

- Inventory analysis: Inputs from the environment (resources and energy) and outputs (product, emissions and waste) to the environment are quantified for each process and then combined in the process flow chart. Allocation of inputs and outputs needs to be clarified where processes have several functions (for example, where one production system produces several products). In this case, different process inputs and outputs are attributed to the different goods and services produced. An extra simplification used by LCA is that processes are generally described without regard to their specific location and time of operation.

- Impact assessment: The effects of the resource use and emissions generated are grouped and quantified into a limited number of impact categories which may then be weighted for importance.

- Interpretation: Interpretation of results in the light of the goal and scope and inventory is critical and sensitive for LCA research. Importantly, the conclusions and recommendations from LCA research should not be extended beyond the project scope.

Agricultural systems have some unique properties that require careful treatment within LCA. In particular, the long production cycle and open system complicate collection of production data and environmental impact data. While these issues are not new to researchers in the agricultural sciences, the interdisciplinary nature of LCA research means careful attention must be directed to the methods and assumptions used during the research.

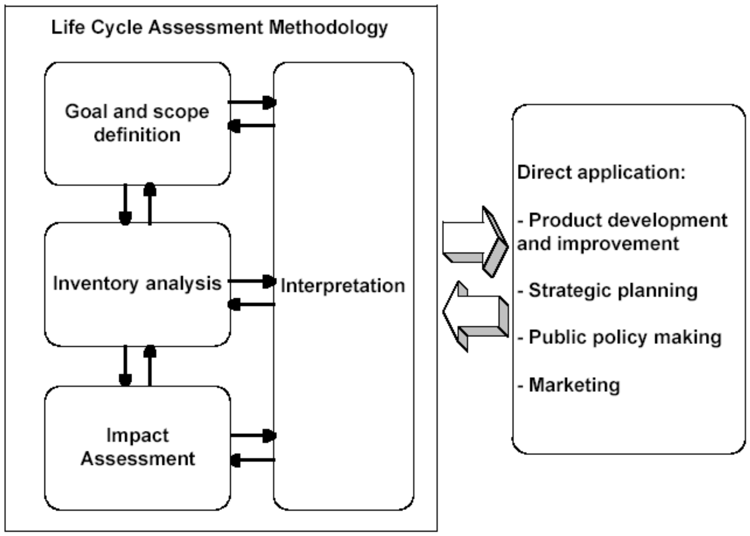

FIGURE 1 – GENERAL FRAMEWORK FOR LCA AND ITS APPLICATION (ISO 2006A: 14040)

LCA may be classified as an applied research tool. This means LCA research does not generally involve conducting individual research studies into each impact area associated with the system. Instead, LCA draws from other studies that have been completed in the area, and relates the results to the system being investigated. Where knowledge gaps exist, the LCA practitioner can either conduct a very brief investigation with the aim of determining how significant the contribution may be from the unknown process, or exclude the process until further research has been undertaken. There are strengths and weaknesses with this type of applied research. One strength is that an LCA can develop broad answers long before the detailed research is completed. A second strength is that the broad scope (i.e. all greenhouse gases associated with a production system) allows impacts to be 'classified' in terms of their overall impact. Likewise, mitigation strategies can be evaluated in a holistic manner. This is something that many scientific research programs find difficult to achieve.

The weakness of an applied research tool such as LCA is that it relies on results from external research and modelling, which is less precise than if a full measurement campaign was done. Modelling or the extrapolation of other research findings can introduce a source of error if there is a significant difference between the conditions of the research and the conditions investigated in the LCA.

It is common for a single product (such as beef) to involve over 2000 processes within the LCA model, consequently the process data used for common products (such as diesel or urea for example) are drawn from Australian and sometimes international databases. A distinction in LCA is made between *foreground data* (or data collected as part of the project from the industries involved), and *background data* (which is drawn from databases or literature sources).

LCA is a complementary tool that can be used in conjunction with detailed scientific R&D. For example, LCA can be used at the beginning of an R&D program to identify the most effective research directions and the potential trade-offs involved with mitigation techniques. Likewise, LCA may be used to evaluate the effectiveness of current research results by bringing them into the context of production systems. As an example of this, LCA can contribute to enteric methane research by addressing a question such as:

Will feeding oil supplements (a strategy that can reduce herd enteric methane emissions per unit of production) reduce net emissions, or will the reduced methane emissions be offset by emissions associated with the production of the supplements?

This is important if real gains are to be made without the fore-mentioned ‘burden shifting’.

2.1.2 Consequential and Attributional LCA

There are two basic perspectives that an LCA study can use. Most LCAs are done retrospectively. This is termed an attributional study, because the impacts are attributed to the product being investigated. The main question for an attributional LCA is “What was the impact of creating this product?” If a study is investigating production for a whole state or nation, every type of system that is currently being used needs to be included to get an accurate and representative result.

An alternative approach is to consider a dynamic system, and investigate the consequences of a change in production. In this case the question might be “what impacts would be created if one more unit (i.e. kilogram) of this product were produced?”

While the attributional study is relatively straight forward to explain, the consequential approach can be more difficult. A consequential study is focused on the marginal production system, i.e. the system that *would be used* if the industry expanded. This is quite an important difference to ‘average production’ and may lead to quite different results. This is particularly important where major technological or geographical shifts have occurred in the industry. Importantly, results from a consequential study cannot be used to comment on the current industry or compared with attributional studies without clear explanation of the differences involved.

The present study took an attributional approach in order to provide a benchmark for the industry across a number of different production systems and states.

2.1.3 Important Methodological Aspects of LCA research

2.1.3.1 Functional Units and System Boundaries

The functional unit in LCA is a measure of the function of the studied system, which provides a reference to which the inputs and outputs can be related (ISO 2006a). This enables comparison of two different systems. For agricultural products, there are three main types of functional unit that can be used. These are mass (kg product), area (ha) or some measure of product quality (e.g. kg protein). The choice of functional unit is particularly important when comparing different systems.

System boundaries determine which unit processes are included in the LCA study. In LCA methodology, all inputs and outputs from the system are usually based on the ‘cradle-to-grave’ approach. This means that inputs into the system should be flows from the environment, without any transformation from humans. Outputs from the system should only occur after all processes (including waste treatment) have been accounted for, so that no subsequent human transformations occur (ISO 2006a). Each system considers upstream processes with regard to the extraction of raw materials and the manufacturing of products being used in the system and it considers downstream processes as well as all final emissions to the environment. Defining system boundaries is partly based on a subjective choice, made during the scope phase when the functional unit and boundaries are initially set.

2.1.3.2 Inventory Development

An LCA study is built on data collected in the inventory stage. For the system being investigated, the inventory covers all inputs (i.e. purchased materials and products, and resources from nature) and outputs (products, by-products, wastes and emissions) for each stage within the supply chain. For industrial systems, collecting inventory data may be relatively simple because the inputs and outputs are relatively static and measured. Generally the focus is on ensuring the data are representative and collecting a large enough sample from the industry being studied to ensure a robust result.

The inventory is typically divided into two different sections: a foreground and a background system. The foreground system represents the part of the system where data are directly collected, and includes:

- production data (i.e. livestock numbers, growth rates, sale records)
- financial (purchases) data (i.e. electricity consumption, quantity of supplements purchased)
- specific environmental data (i.e. water usage, vegetation management, soil management, analyses etc.).

The background system covers other elements of the supply chain where data was not collected directly from businesses but were accessed from databases or modelled.

For agricultural systems, two main differences exist compared to industrial systems. Firstly, production may not be static from year to year, and secondly, some inputs and outputs are very difficult to measure. Consequently, the inventory stage of an agricultural LCA is far more complex than most industrial processes, and may require extensive modelling in order to define the inputs and outputs from the system. For this reason agricultural studies often rely on a far smaller sample size and are often presented as ‘case studies’ rather than ‘industry averages’. For agricultural systems, many foreground processes must be modelled or estimated rather than being measured. Assumptions made during the inventory development are critical to the results of the study and need to be carefully explained in the methodology of the study. In order to clarify the nature of the inventory data, it may be useful to differentiate between ‘measured’ and ‘modelled’ foreground data. For a cattle business, measured foreground data would include fuel use and livestock numbers, while modelled foreground data would include enteric methane emissions.

2.1.3.3 Handling Co-Production

Most production systems produce both primary and secondary products. Within LCA, there must be some means of dividing the impacts between these multiple products. This process is very important and can have a large bearing on the result.

The beef production system has a number of co-products or potential co-products across the supply chain, depending on the perspective taken. For example, cull cows may be considered a co-product of prime beef production. This perspective would be based on differences in the quality of the two products. However, a number of difficulties exist with this perspective. Firstly, the difference in quality is not uniform. Some beef from cull cows (sirloin etc.) may be sold into the fresh meat market because the quality is sufficient. Secondly, the choice here makes a value judgement based on product quality rather than nutritional value. From a nutritional perspective, there is no reason for differentiating between beef from cull cows that is used for mince and beef used for steak. Here it can be seen that choices relate to the perspective of the study.

A second potential co-product from beef production arises from the feedlot. Feedlot cattle manure is a low value by-product that is typically spread on crops or pasture as a fertiliser replacement. While some may consider this a waste (and therefore not a co-product), it is not considered this way by the industry. Consequently, this must be addressed within a project.

The clearest 'primary product/co-product' examples arise at the point of slaughter. Examples are hides, edible and non-edible offal, tallow and meal products. The approach used for handling these can have a large bearing on the impacts attributed to beef post slaughter.

The options for handling co-production according to ISO 14044 (ISO 2006b) in order of preference are:

- Clear subdivision of the system, or system delineation.
- System expansion (expanding the product system to include the additional functions related to the co-products to avoid allocation).
- Allocation on the basis of physical or biological relationship (mass or energy for example).
- Allocation on some other basis, most commonly economic (market) value.

The choice of method for handling co-production can have a large impact on the results. This is discussed in detail in the methodology section.

2.2 Australian Agricultural LCA Research

2.2.1 Current and Previous Australian Beef LCA Research

Meat and Livestock Australia Ltd (MLA) has funded a number of LCA projects in the grazing beef sector over the past six years. Completed studies include Peters et al. 2010a, b, 2011), Eady et al. (2011), Ridoutt et al. (2012) and Ridoutt et al. (2011). Each of these studies included GHG emissions and water use, while Peters also included energy use and nutrient management. Feedlot LCI projects have been completed by Davis and Watts (2006) and Davis et al. (2008a, 2008b). This LCI work was expanded in 2010 by Wiedemann et al. (2010c).

In order to understand the comparability (or otherwise) in these studies, five critical assumptions were reviewed and are presented in Table 1. To clarify the methods used for handling co-products used in previous MLA funded research, Table 1 shows these, with a standard value for GHG as an example.

TABLE 1 – REVIEW OF PROJECT ASSUMPTIONS FOR AUSTRALIAN BEEF LCA STUDIES

Reference	System boundary	Method for handling co-products	Method for estimating GHG and water	Functional Unit
MLA project FLOT.328 (Davis & Watts 2006), MLA project B.FLT.0339 (Davis et al. 2008a, b)	Feedlot gate to gate	All impacts allocated to beef – same as 'unallocated'	GHG estimated using DCCEE methods with livestock performance data. Water use measured.	Kg of HSCW gain at the feedlot. Use of a carcase weight unit implied some approach to handling co-products (meat, hides etc.). However, this was not completed. The impacts were all directly attributed to the meat product.
COMP.094 (Peters et al. 2010a)	Nominally included all impacts through to (and including) meat processing. However, results for the Victorian supply chain were reported in one year (2002) without including the impacts of cattle breeding.	Mass allocation of impacts at the point of slaughter	GHG estimated using DCCEE methods with livestock performance data. Water use estimated using a farm hydrology model.	Kg of HSCW at the meat processing gate. HSCW was selected because it is a common industry unit. However, it does not accurately align with the production system (i.e. HSCW is rarely the output of a meat processing plant).
Eady et al. (2011)	All impacts through to the farm gate.	Allocation between cull cows and slaughter cattle done on an economic basis.	GHG estimated using DCCEE methods with livestock performance data.	One kg of prime cattle live weight (either weaners or slaughter cattle) at the farm gate.
Ridoutt et al. (2011, 2012)	All impacts through to the farm gate.	Not clear.	Water use was predominantly modelled from livestock data and literature assumptions.	One kg of prime cattle live weight (class of cattle depended on the case study) at the farm gate.

2.2.2 LCA Methodology Development

Methodology development for LCA in Australian agriculture was enhanced by the funding of a LCA methodology project coordinated by the RIRDC (Harris & Narayanaswamy 2009). This project focused on GHG, energy and water assessment. In general this document represents a slight refinement of the international standards (ISO 14040-14044) with some specification regarding on-farm data collection and the handling of water.

3 Sustainability in the Beef Industry

3.1 Introduction

The 'sustainability' of food production systems is bounded by the constraints of renewable resource supply, maintenance of natural capital and ecosystem function, and maintenance of 'services to humanity' which include both food/fibre production and production of clean air, water etc. Producing beef in a sustainable production system is a high priority for the beef industry. However, "sustainability" is a broad term with numerous separate elements, making it far from simple to define or achieve in practice. Sustainability has been broadly defined as "ecological stability, economic viability and socio-cultural permanence" (Lal 1991). For Australian agriculture, the SCA define sustainability as '*the use of farming practices and systems which maintain or enhance the economic viability of agricultural production; the natural resource base; and other ecosystems, which are influenced by agricultural activities*' (SCA 1991). Although these concepts are not new, few studies have attempted to quantify the sustainability of the Australian beef industry in a holistic manner.

Fundamentally, the sustainability and stability of an industry (or society as a whole) rests on maintenance of natural capital (Goodland 1995). Social and economic sustainability is not possible if the resource base is no longer able to produce food. Hence, agricultural sustainability is not simply an issue for agricultural industries, but for society as a whole. This has been highlighted by recent attention to global food security, which must be underpinned by sustainable agriculture (UNEP 2012). Food production is increasingly being seen as a critical issue for the next century, with the FAO (2009a) predicting that world population will increase by 34%, with a corresponding increase in demand for cereal grain (43%), and demand for meat (74%). Increased demand for food will place greater pressure on limited land resources (particularly arable land) and on competition for commodities such as cereal grain that can be directed either to meeting human food requirements, or indirectly to livestock. The disproportionate increase in the demand for meat is expected as a result of rising incomes, resulting in a shift from plant protein sources to animal protein sources. Australia, as a major global exporter of red meat (beef and sheep meat) and grain (predominantly wheat) has an important role to play in maintaining and increasing the supply of primary food available for global trade and thus contributing to food security in nations that are net food importers.

The focus of the present study is on the fundamentals of environmental sustainability in the beef industry, taking into account the key role that agriculture has in producing food for the world. The key elements of the investigation are therefore:

- Utilisation efficiency of key natural resources such as land, water and energy.
- Assessment of environmental impacts on land, water and air quality.

In theory, natural resources are renewable and may be used indefinitely provided they are maintained and not overstretched. However, the supply of these resources at any given time is finite, and consequently the temporal availability and efficiency of use is highly relevant, particularly in the context of increased demand for food production worldwide. Where non-renewable resources such as fossil fuel energy are used, sustainability in the long term will be constrained by the availability of these resources, and utilisation efficiency is a key measure of sustainability in the short-medium term.

Environmental impacts inevitably arise from production systems as a result of general operations. These impacts may damage any or all of the following; the resource base, the health of natural ecosystems or human health. In some instances the cause-effect relationship is clear. For example, phosphate losses from a farm can cause eutrophication (elevated nutrient levels) in a local river, leading to declining aquatic ecosystem health, changes in fish species or fish deaths. This may happen rapidly (i.e. in the space of months or years) and the result of improved practices may also be seen rapidly. On the other hand, the impacts of greenhouse gas emissions from a farm are less easily conceptualised. These impacts contribute to a global phenomenon with numerous causes and uncertain effects. Additionally, there is a very weak link between cause and effect at the local level, making it hard to 'see' the impact of emissions from a given farm. None the less, such assessments must be made, because agriculture can have a significant contribution to overall impacts when whole industries (rather than individual farms) are taken into account. These aspects of environmental sustainability are shown in Figure 2.

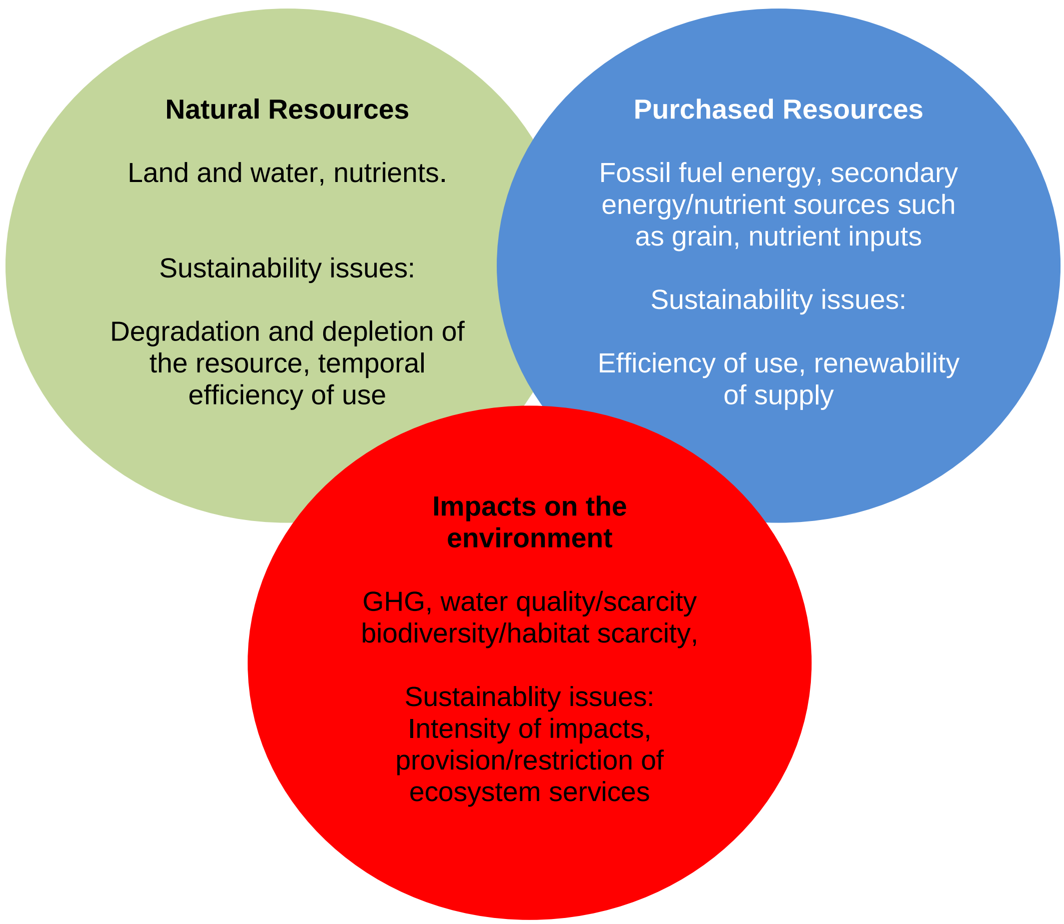

FIGURE 2 – KEY ELEMENTS OF AGRICULTURAL SUSTAINABILITY RELATED TO RESOURCE USE AND THE ENVIRONMENT

The following sections provide a discussion of these three broad areas with respect to Australian beef production.

3.2 Resource Use

3.2.1 Land occupation

Land resources are a limited global resource. Globally, of the total ice-free land surface of 13.4 billion hectares, approximately 3.5 billion ha (27%) are permanent pastures and 1.5 billion ha (12%) are under cultivation (arable). With a growing demand for food and biofuel production from the world's land resources, utilisation efficiency is an increasingly important factor, though there is a general lack of consensus on how this should be measured in LCA. To date, most assessments have reported simply the total land required by a production system (i.e. for beef or pork or wheat) with no description of the type of land used, or the impact of using that land. Land types differ in productivity and suitability for cultivation and this needs to be taken into account in order to provide meaningful results.

It has been estimated that while an additional 2.8 billion ha is potentially arable, if natural restraints are taken into account, a more realistic estimate is around 1.5 billion ha (Bruinsma 2009). Even to realise a doubling of the area currently under cultivation would require a marked acceleration in investment in capital and infrastructure, construction and possibly reclamation. In fact, FAO data show that the net increase in arable land has been only 5 million ha per year over the past two decades and the likely further increase is more likely to be about 5% (rather than the 50% suggested by Bruinsma 2009) by 2050 (FAO 2009b). The potential for increase in arable land is even more restricted in the developed countries and will likely decline.

Of the total land area of Australia (7.687 million sq. km) only 7% is arable according to the (FAO 2008). However, at any given time closer to 3% is actually cultivated (BRS 2010). Considering there are state regulations restricting conversion of pasture land to crop land, the total arable land may be closer to 3% than 7%. In contrast approximately 56% of Australia's land area is used for grazing livestock, mainly on native or naturalised pastures (Figure 3).

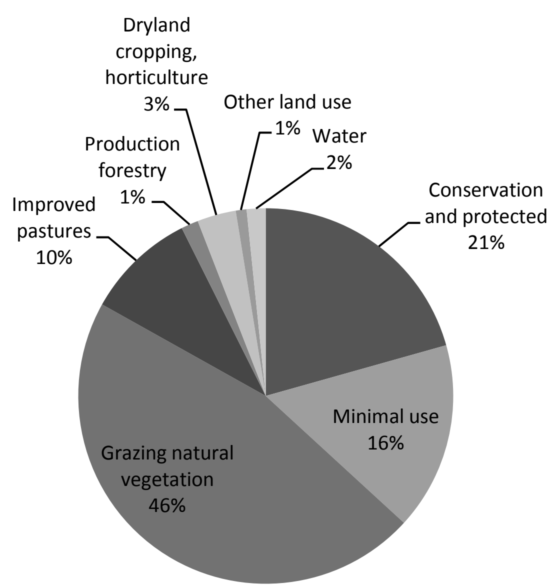

FIGURE 3 – MAJOR LAND USES IN AUSTRALIA BASED ON THE 2005-06 DATASET (BRS 2010)

The vast majority of grazing land falls in the pastoral zone, which is generally unsuitable for other forms of agricultural production, particularly those reliant on cultivation, because of land and climate limitations. Land in the category “improved pastures” may be a combination of arable and non-arable land. However, because of regulatory constraints in some states (such as NSW), much of the pasture land that could be cultivated (from a land capability point of view) is restricted from conversion by legislation. In Australia, arable land used for cropping represents only 3.4% (0.26 M ha.) of total land mass. Consequently, this is a a much more limiting resource and is subject to a much higher degree of competition for food production uses. The dominant competitive agricultural users for arable land in Australia are grain (cereal and pulse) production, forage (crop) production for grazing animals and pasture production for grazing animals. It is informative therefore to investigate land occupation for different livestock systems in terms consistent with land capability and availability. While incomplete, it appears necessary to distinguish between arable and non-arable land types *at a minimum* when assessing land occupation from a resource perspective.

There is potential to convert land from one land use to another, though this is constrained by land type (soil, slope etc.), vegetation, annual rainfall, rainfall variability and evaporation. Land use mapping by the Australian Bureau of Rural Sciences (BRS 2010) shows that in the five year period from 1996/97 to 2001/02, the area of land with natural vegetation used for production fell by 12.7 million ha. This was due to an 11.6 million ha. decline in grazing land. Approximately half of the rangelands lost from production were converted to cropping and half to conservation reserves. More recent statistics from the Australian Bureau of Statistics show the area under crops and the protected land area has continued to increase while non-crop farm area (predominantly grazing) has declined (Figure 4). The trend towards taking land from production to conservation is likely to increase. For example, in 2009 the Queensland government announced as part of the State’s climate change policy that there was an objective to increase the protected area from 8.3 M ha. to 20 M ha. by 2020.

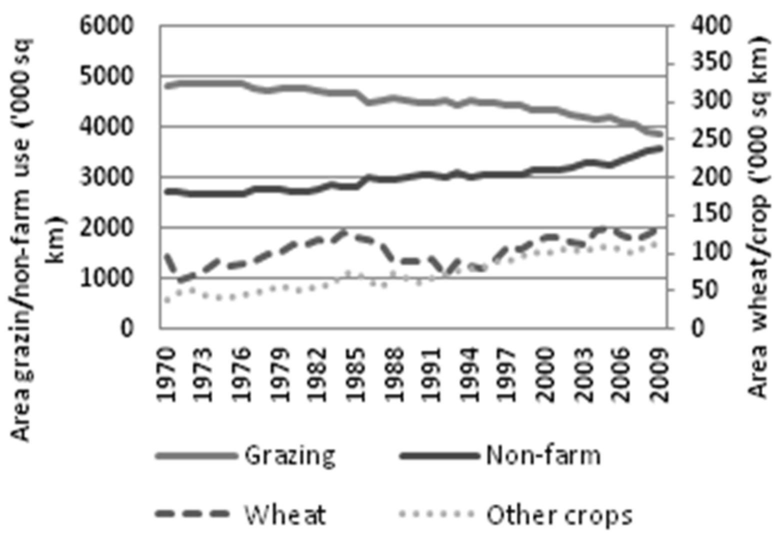

FIGURE 4 – TRENDS IN LAND USE FOR MAJOR AGRICULTURAL PRODUCTION IN AUSTRALIA (ABARE 2009A)

Future climate change may reverse the trend towards increasing areas under cultivation with some predictions indicating that lower effective rainfall will drive conversion of more marginal croplands to permanent pastures (PMSEIC 2010). The potential for expansion or intensification of productive rangelands has also been affected by legislation by State governments to end broad scale land clearing in the past two decades, in particular in New South Wales and Queensland. Vegetation management policies may also affect the potential for sustainable intensification of production in savannahs through restrictions on clearing to manage woody encroachment, regrowth and woody thickening. Stopping broad scale clearing using chemical or mechanical methods to manage woody regrowth and thickening or to offset the impact of woody proliferation by clearing remnant woody vegetation is predicted to move current tree/grass balance away from grasses and have a negative impact on livestock carrying capacity (e.g. Burrows et al. 2002).

3.2.1.1 Land occupation Assessment in LCA

To date, land occupation has most commonly been reported using a simple estimate of 'total land occupation' over a given time period, measured in square metres (m^2 yr). Examples from beef LCA studies are provided in
Table **2**. The extensive review of beef, pork, chicken, egg and milk LCA studies by de Vries & de Boer (2010) showed that beef production requires the greatest amount of land of all the livestock protein products, which is not surprising considering the differences in fecundity and feed conversion efficiency between the species. The authors note that the analysis is insufficient to recommend a shift from red meat to white meat because the land resource utilised by each is quite different: ruminants can graze non-arable land, while non-ruminants require grain grown on arable land. They also note that poultry and pigs require grain which could be fed directly to humans, while red meat production may not. This should be seen as a major limitation to the usefulness of the findings.

Recent advances in land occupation methodology recommend that in addition to the area used and the duration for which it is used, there should be an assessment of the change in land quality caused by using land (Mila i Canals et al. 2007). In the present study, we chose to separate 'land occupation' (as a measure of resource utilisation) and 'land occupation impacts' (as a measure of the change in land quality as a result of use). These are closely aligned and may in the future be integrated into a single measure.

Progress in refining the land occupation assessment is currently progressing in two directions. One approach would be to disaggregate land into a number of capability classes (arable, non-arable, irrigated arable etc.). The second would be to apply a weighting factor in order to standardise the measure of land occupation against land productivity. The primary approach suggested here is to use Net Primary Productivity (a measure of biomass accumulation, most commonly measured in units g C m^2yr) to 'weight' land occupation against a standard reference (i.e. a national or global average). We have taken the first approach in the present study, though this may need to be refined by future methodology development.

TABLE 2 – LIFE CYCLE LAND USE (OCCUPATION) FOR BEEF PRODUCTION PER KILOGRAM OF LIVE WEIGHT PRODUCED

Reference	Country	System	Land Occupation m^2 yr/kg LW
Williams et al. 2006	England and Wales	Beef sourced from dairy calves and purpose-grown beef herds	12.7
		Beef sourced from purpose-grown beef herds	21.2
Pelletier et al. 2010	USA	Calves backgrounded on wheat pastures and finished in feedlot	84
		Calves finished on managed pasture and hay	120
Nguyen et al. 2012	France	Four pasture based beef production systems using different feeding strategies	26.1 (25.9-26.4)

3.2.2 Water Use

Stress on fresh water resources is a growing concern both in Australia and globally. The World Health Organisation have estimated that 1.1 billion people do not have access to improved water supply sources (WHO 2009). With a growing human population, it follows that stress on water reserves will increase dramatically in the next 30-40 years (Rockström et al. 2007). While water scarcity is a relatively difficult term to define, there is little doubt that water resources are under considerable pressure worldwide (Falkenmark et al. 1989, Glieck et al. 2009, Shiklomanov 1998). Agriculture is attributed with using 65-70% of water extracted from the environment in Australia (ABS 2006), which is similar to the situation globally. Of the water used for agriculture, most is used for irrigation, with smaller amounts used for livestock.

The ABS reports one category that is specifically related to beef (irrigation water used for grazing meat cattle). Some other categories may contribute to water use in the supply chain (i.e. for the production of feed inputs for grazing or lot feeding). The ABS does not collect data relating to on-farm dams used for livestock drinking water and does not take into account drinking water from creeks or rivers. It is possible some bore water used for drinking is included in the data; however for all intents and purposes; cattle drinking water is excluded from the ABS data. Australian water use data for a number of agricultural industries are presented in Figure 5.

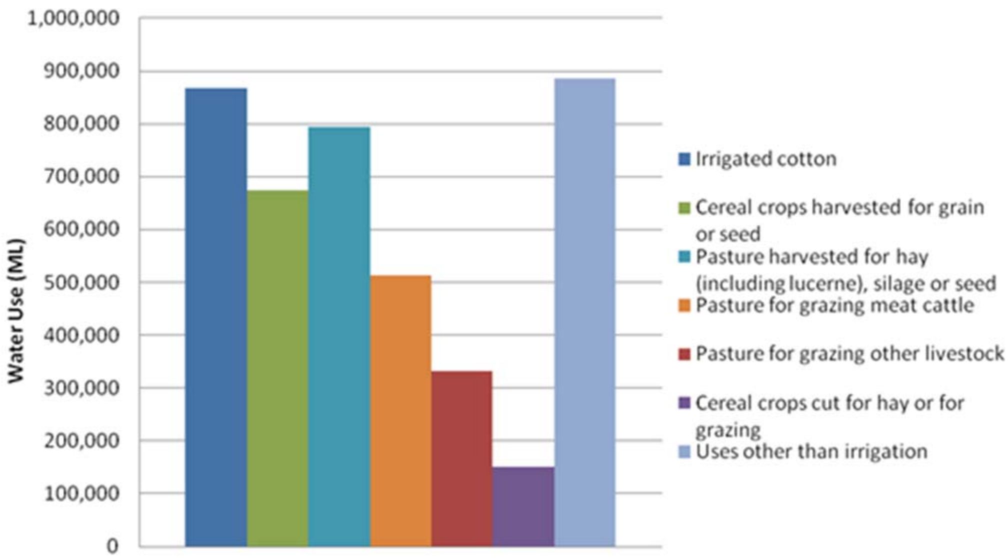

FIGURE 5 – WATER REQUIREMENTS FOR A NUMBER OF AGRICULTURAL COMMODITIES (ABS 2008)

While Australia has adequate water resources nation-wide, not all water resources are easily accessible to areas of high demand, and competition for water resources is one of the most severe resource allocation issues facing the country.

Water 'use' is an ambiguous term that may include both consumptive (i.e. evaporative) and non-evaporative uses (i.e. cleaning water that is then released to the environment). Evaporative use or water consumption directly limits short term availability to other users. While evaporated water eventually returns via precipitation, the timing and distribution of rainfall is variable, hence the two should be differentiated. This requires use of a water balance at different stages in the supply chain in order to determine the volume of water extracted and the amount subsequently released (Bayart et al. 2010). Non-evaporative uses may be classified based on their suitability for different purposes (Boulay et al. 2011). It is important to note that, where water flowing from a system is degraded in quality but is still suitable for other users, it may be considered a flow rather than a use, despite a change in quality. However, uses that result in degradation of water quality should be clearly described.

Another agricultural water use issue relates to the relationship between land occupation and impacts on the natural water balance. Many agricultural systems modify the water balance by changing the proportion of rainfall runoff from an area of land. In such situations, Mila i Canals et al. (2009) suggests that differences in the water balance between the current land occupation and the 'reference' land occupation (i.e. open forest etc.) be attributed to the system. Interestingly, Mila i Canals et al. (2009) considers 'pasture and meadow, extensive' land occupation with <600 mm rainfall/yr to have a higher evapo-transpiration rate than the reference land occupation (forest). This is not accurate for most regions of Australia, where clearing of native vegetation has resulted in higher runoff (Brown et al. 2005). For heavily transformed land occupation (i.e. industrial areas, roads etc.) Mila i Canals et al. (2009) classifies runoff as 'lost' water. While the application of this may be reasonable for some industrialised settings where runoff cannot be utilised (because of contamination etc.) it does not appear to be a universally applicable assumption. Feedlot beef production provides a useful agricultural case study, as the feedlot is a highly modified land occupation that increases runoff significantly (Lott 1994) and results in degradation of water quality because the runoff from the cattle pens collects an amount of manure, containing nutrients, salts, organic material and possibly pathogens. However, feedlots are constructed in such a way that effluent is treated to reduce the organic load, and water is then available to be utilised for crop production under specific guidelines (Skerman 2000). In this situation, the feedlot dramatically increases the volume of runoff from the area compared to the reference situation, but this water is carefully managed to ensure it does not contaminate the environment. This is done via on-site irrigation of crops (usually hay or silage crops which are then fed back to the cattle in the feedlot). The net change in the water balance from the feedlot property (the feedlot catchment and the irrigation area) is generally either positive (runoff is increased, albeit of lower quality) or the balance is relatively static because runoff water is increased from the feedlot area, stored and then irrigated onto crops where almost all is lost to the atmosphere via evapotranspiration. In this situation, consumptive water use should be considered as the difference between runoff in the reference situation and the occupied land use. Further details regarding inventory methods for determining water use in LCA are documented in Appendix 3 – Water Use Inventory.

3.2.2.1 Virtual Water and Water Footprinting

The discussion of water use for livestock production has been complicated in recent years by the use of the virtual water (VW) and water footprint (WF) concepts. These arose independently of LCA and were used originally as a means of describing the water required to produce tradable commodities (particularly food) in water stressed economies (Allan 1998). The VW method makes a useful contribution to the global understanding of water transferability by showing that irrigation water in one region can be saved by importing food, thereby reducing water stress. Moreover, stress on irrigation water because of agriculture can be alleviated by growing products in regions where water requirements can be met from rainfall rather than from irrigation.

To further improve the understanding of VW, Falkenmark (2003) introduced the terms of ‘blue’ water (which represents our general understanding of freshwater resources from surface or groundwater supplies) and ‘green’ water, which may be classed as evapotranspiration water (i.e. Falkenmark 2003, Falkenmark & Rockstrom 2006) or ‘soil stored moisture from rainfall’. A third term ‘grey water’ was added to describe the water requirement for assimilating pollutants from a system. All three of these terms are now used in the field of water footprinting (Hoekstra et al. 2009a, Hoekstra et al. 2009b)(and Hoekstra et al. (2011).

The key difference between an assessment of ‘water use’ for livestock production using the traditional understanding of water (essentially blue water: water extracted from rivers, dams, lakes and aquifers) and the VW/WF concept relates to the inclusion of rainfall for growing plants used to feed livestock (green water), and water used to assimilate contaminants released from the system (grey water). Green water ‘use’ by livestock systems is very large (>98% - Peters et al. (2010b)), which results in very high estimates of VW/WF for livestock products compared to estimates of extracted or consumptive fresh water only (see Table 3). However, inclusion of green water is not generally relevant to an assessment of the impacts of water on either competitive users or the environment. Where the purpose of the study defines water use and impacts in terms of competitive users (i.e. agricultural water use, industrial water use, domestic water use) and the environment (aquatic ecosystems) then green water is not relevant.

Grey water is also a complicated term. The water required to assimilate contaminants released by a production system is essentially the investigation of secondary causes. The concern in each instance is the amount of contaminant released. In LCA, this is addressed directly by using indicators such as eutrophication. The second issue with defining grey water in agricultural systems relates to the classification of water use. Where water is ‘contaminated’ with nutrients, this is of no concern to most agricultural water users, because nutrients are only considered a contaminant when the water is to be used for some industrial purposes, domestic purposes or release to the environment. Hence, calculation of grey water would need to be location specific, based on the release limits for key 'contaminants' in agricultural water.

3.2.2.2 Water Use for Beef Production

TABLE 3 – LITERATURE ESTIMATES OF 'WATER USE' REQUIRED TO PRODUCE ONE KILOGRAM OF BEEF

Water Use (L/kg LW)	Methodology	Functional Unit and System Boundary in original study	Country	Reference
Virtual water / Water footprint				
56,000 [a]	Not defined by author	Unclear – Pasture and grain fed cattle, likely to include upstream impacts from breeding	USA	Pimentel et al. (1997)
8,000 – 37,000 [a]	Not defined by author	1 kilogram of meat, Boundaries are unclear	not known	Gleick et al.(2009)
23,000 [a]	Not defined by author	Unclear – grain fed cattle.	USA	Pimentel et al. (2004)
9,000 [a]	Virtual water / water footprint – methodology defined	Boneless beef (excluding impacts from breeding herd)	Australian average	Hoekstra and Chapagain (2007)
8,000 [a]	Virtual water / water footprint – methodology defined	Boneless beef (excluding impacts from breeding herd)	World average	Hoekstra and Chapagain (2007)
7,451-12,855	Water footprint (green + blue water only)	Live weight	Two Queensland farms	Eady et al. (2011)
Extracted water / Consumptive water use (LCA – inventory results)				
30-405	Extracted water use - LCA	Hot Standard Carcase Weight – supply chain to meat processing	Two Australian supply chains	Peters et al. (2010b)
24.7-234	Consumptive fresh water use	Live weight – supply chain to farm gate	Six Australian supply chains	Ridoutt et al. (2012)
51.1-155	Blue water use	Live weight – supply chain to farm gate	Two Queensland farms	Eady et al. (2011)

[a] Water use estimate converted from carcase weight to live weight using a conversion factor of 0.53 in the absence of specific data from the study to enable the conversion.

The purpose of LCA is to investigate not simply the 'use' of a resource, but to determine the potential impact of that use. This is important for the discussion of water use. Consumptive water uses vary in their impact on other competitive users or the environment. Where water is plentiful, the relative stress on water reserves may be very low. Put simply, the '*the more you use, the worse you are'* principle is not universally applicable comparison of water use between different catchments. Consequently, the impact of using water may also be low, either on other competitive users (because there is plenty to go around) or the environment (because there is sufficient water to maintain aquatic ecosystem health at the current level of abstraction). To address this, impact assessment methods have been proposed by Mila i Canals et al. (2009) and Pfister et al. (2009). Pfister et al. (2009) described a method of determining the 'stress weighted' water use, by accounting for the expected impact of using water in a given catchment, using a global stress weighting factor. Ridoutt & Pfister (2010) further describe this method and apply the term ‘stress-weighted water footprint’, with units of L H_2O-e. The stress weighted water use impact assessment method applied different stress weighting factors for different regions of Australia. To calculate the stress weighted water use, consumptive water use 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 (H2O-e; Ridoutt & Pfister 2010). Using this approach, Ridoutt et al. (2012) estimated that the stress weighted water use for beef produced from a number of NSW production systems ranged from 3.3 – 221 L H_2O-e / kg LW. We applied the same method in the current study.

3.2.3 Energy Demand

Fossil fuel energy inputs are essential to agricultural production. Energy is required in the grazing sector to pump water, operate agricultural equipment (tractors, harvesters etc.) and vehicles, and for mustering livestock. The majority of this energy requirement is met using combustible petroleum based fossil fuels (diesel) or to a lesser extent electricity. In LCA, energy use is assessed across the whole supply chain, where the largest sources of energy use often arise from farm inputs such as fertiliser or feed, rather than direct use of diesel or electricity.

Assessment of energy use (generally termed ‘energy demand’) generally includes energy sourced from fossil and non-fossil sources, but does not include energy digested by animals. Energy use is less commonly assessed than GHG or water use. Our review of the literature only identified two studies in addition to the previous study by Peters where energy use was reported.

3.2.4 Grain Use – Human Edible Protein and Energy

Grain is an important primary commodity which can be used either for human consumption or animal production (and subsequent human consumption of animal products). Australia is a major global grain producer and exporter. However, domestic consumption has increased rapidly over the past 10 years, primarily driven by increased consumption from livestock production (Spragg 2008). Livestock consumed an estimated 28% of grain produced in 2007 (Spragg 2008). The use of cereal grain for livestock feeding is important both from an environmental impact and a food security perspective, and is an important focus for research in both areas. Because grain can be used directly for human consumption, there is a potential conflict between livestock production and food security where livestock are fed grain. However, this must be balanced against other factors influencing food security such as consumer preferences and beneficial nutritional characteristics of animal proteins. It must also be considered when assessing environmental impacts.

The efficiency with which animals convert feed into product (termed the feed conversion ratio, or FCR) is a very important performance indicator for all livestock systems. There are marked differences between the species in terms of FCR; poultry are the most efficient, followed by pigs, then ruminants (cattle and sheep). Differences between the species arise from fundamental physiological differences. In particular, monogastrics (poultry and pigs) have a much more efficient digestive system for high starch (grain) diets. The monogastric species also have higher fecundity (more offspring per breeding animal) resulting in lower maintenance feed requirements for the breeding herd or flock. For example, breeding sows consume in the order of 55-65 kg feed / weaned pig, and produce 20-24 sale pigs per sow per year (see Wiedemann et al. 2012a). In contrast, a beef cow may consume 3500 kg of feed per calf produced. It is also typical for beef herds to produce fewer than one calf per cow on average across a herd. At 75% weaning, the breeding cows will consume 4700 kg of feed per calf weaned (not accounting for the feed consumed by the calf). However, one very important difference exists. Beef and sheep consume grass, which has a very low level of digestibility for monogastric animals. Consequently, the whole herd/flock FCR is not comparable between poultry/pigs (which consume mainly grain diets) and sheep/cattle, which consume mainly grass diets. Where ruminants are fed grain (i.e. lot feeding) the comparison is more meaningful, because the FCR when consuming grain is still much lower than monogastric species.

CAST (1999) reported the ratio of human edible energy and protein consumed by livestock species compared to the amount produced as a way of quantifying the contribution or conflict between animal production and food supply. This metric, which could be termed the 'human edible feed conversion ratio or H-FCR' of a livestock system is informative to the discussion of animal agriculture's contribution to food supply. Gill et al. (2010) noted this was an important factor in the discussion of livestock's role in mitigating climate change in the context of food security. The human edible protein and energy FCR for a number of species were reported by Gill et al. (2010) citing CAST (1999). These results are reproduced in part in Table 4.

TABLE 4 – COMPARATIVE EFFICIENCIES OF DIFFERENT LIVESTOCK PRODUCTION SYSTEMS IN TERMS OF HUMAN EDIBLE ENERGY AND PROTEIN (REPRODUCED FROM GILL ET AL. 2010)

	Energy				Protein			
	USA		South Korea		USA		South Korea	
	Total efficiency	Human edible efficiency	Total efficiency	Human edible efficiency	Total efficiency	Human edible efficiency	Total efficiency	Human edible efficiency
Beef	0.07	0.65	0.06	3.34	0.08	1.19	0.06	6.57
Pigs	0.21	0.31	0.2	0.35	0.19	0.29	0.16	0.51
Poultry Meat	0.19	0.28	0.21	0.3	0.31	0.62	0.34	1.04

Table 4 shows the higher human edible conversion efficiency of South Korean production, because of the higher use of forages rather than grain (for beef) compared to the USA.

Environmental impacts from livestock production systems are also related to grain use. Therefore, grain use and associated impacts must be taken into account in an analysis of the impacts of livestock on sustainability and food security.

3.3 Environmental Impacts

3.3.1 Eutrophication Potential

Eutrophication is the process of increasing organic enrichment (via growth of aquatic organisms) in an aquatic ecosystem, leading to ecosystem damage. This is primarily the result of phosphorus and nitrogen export to waterways. The relationship between phosphorus and nitrogen releases and organic enrichment was first established by Redfield et al. (1963) by determining the ratios of carbon, nitrogen and phosphorus in phytoplankton. The so-called Redfield ratio (C:N:P of 106:16:1) is the basis for eutrophication characterisation factors, using phosphate equivalents. Anthropogenic nutrient inputs to aquatic environments can upset the natural balance of supply of nutrient supply and biomass production. This leads to unnaturally high levels of plant production and accumulation of organic matter that degrades water quality and reduces oxygen content, leading to disruptions in the ecosystem of the waterway. Nutrient loss from grazing and cropping land is a frequently-discussed issue of environmental concern.

The conventional understanding of eutrophication suggests that freshwater ecosystems are most strongly P limited, while marine ecosystems are N limited. Consequently, determination of eutrophication potential is dependent on the ecosystems affected. However, the conventional understanding is not universal. Australian research suggests that nitrogen is also limiting in freshwater ecosystems (Davis & Koop 2006).

As noted by Gallego et al. (2010), global or country scale characterisation factors are not sufficient for determining the impact from eutrophication in countries with large geographic and climatic variability. This is acutely apparent in Australia, where the factors contributing to eutrophication are known to vary widely between catchments (Davis & Koop 2006), making global or even country specific characterisation factors inadequate.

Country or regional eutrophication characterisation factors may also incorporate transport factors to determine the proportion of the substance likely to be transferred to the receptor (Gallego et al. 2010, Huijbregts & Seppälä 2001). Such regionally specific characterisation factors have not been developed in Australia to date. Moreover, the state of the science in Australia suggests that there are substantial differences in the cause:effect relationship between sources and impacts for freshwater eutrophication in Australia compared to Europe, and indeed, between one catchment and the next within Australia (Davis & Koop 2006).

While nutrient loss to waterways is a topic of national concern in Australia, there was insufficient primary research available to develop characterisation factors and quantify eutrophication for the supply chains investigated in this study. In lieu of this, a qualitative discussion of eutrophication potential has been included for each supply chain. To provide general context for the discussion, a summary nutrient loss pathways relevant to Australian grazing properties, together with a review of the incidence and causes of eutrophication in relevant catchments, is included below.

Mid-point and end-point eutrophication assessment in LCA requires a strong cause-effect relationship to be established between i) the production system and the source of nutrient losses, ii) the source of nutrient losses and the sensitive receptor (i.e. the river, estuary or ocean), iii) the nutrient source and the impact (i.e. observed algal blooms). Fundamental drivers of eutrophication noted by Davis & Koop (2006) for inland river systems (relevant to the SW supply chain) were as follows:

- Stratification and light penetration, not nutrient availability, are the triggers for algal blooms in major inland river systems of Australia such those found in the Murray Darling Basin.
- Both nitrogen and phosphorus may be limiting to freshwater eutrophication in Australia.
- Diffuse sources dominate total nutrient discharge to waterways. However, total quantity is only one factor controlling ecosystem impact, along with the timing, location and nature of the loading.
- Studies of three major and one minor inland river in the Murray Darling Basin (MDB) showed no trace of fertiliser derived phosphorus. The predominant source of phosphorus is from stream bank erosion processes in this catchment. There is evidence to suggest that erosion rates have been accelerated.
- Loss pathways from the field level to the river are not well understood, and further research is needed to develop suitable transport factors. This is particularly true for nitrogen, which has received less attention than phosphorus.

A second review of nutrient export to waterways in Australia (Drewry et al. 2006) noted that grazing may result in significant losses of nitrogen and phosphorus via overland flow and groundwater pathways at the paddock level. However, these findings were predominantly based on research from southern Australia, and impacts were much more apparent from dairy farming than either sheep or beef cattle grazing. There was agreement between Drewry et al. (2006) and Davis & Koop (2006) that research was required to understand nutrient transport processes to link nutrient source data with receptors. The degree of nutrient saturation in the flow pathway from fields to streams, and the presence of farm dams which may act as nutrient sinks, may influence the nutrient transportation process.

Few studies of nutrient loss were available for the northern, summer dominant rainfall regions of Australia. The summer dominant rainfall zone differs to southern Australia because the period of highest rainfall aligns with the period of highest evapo-transpiration, resulting in soil moisture deficits and low levels of leaching for regions with comparable annual rainfall (see McLeod et al. 2006). While nutrient losses may occur in these regions, the rates are unknown, and unlikely to be reflected by research in southern Australia. The two farms used no fertiliser, and had slight nutrient deficits on the farms. With the relatively low annual rainfall, there is reason to believe the nutrient losses from leaching and runoff would be low, and are excluded from the DCCEE (2010) calculations for the regions where the farms were located. Nutrient loss with soil erosion may be a concern from both these farms, though the contribution this makes to Eutrophication is not easily ascertained. Sediment and nutrient losses in the Burdekin catchment (relevant for the NE farm) are a concern because this water flows to the Great Barrier Reef of the coast of Queensland. Research suggests that these impacts primarily arise from a very small portion of the catchment (Roth et al. 2003) rather than the region where the case study farm was located. The location of the Burdekin dam may also restrict the impact of grazing nutrient and sediment losses to the lower Burdekin and the ocean.

3.3.2 Land Occupation Impacts

We chose to differentiate ‘land occupation’ from ‘land occupation impacts’ – the latter describing processes that result in land degradation and ultimately, land depletion (where land is no longer suitable for agricultural production). Land occupation impacts should also be assessed where land transformation occurs (i.e. changing a pasture to cultivation or vice versa). Land degradation is one of the primary agricultural sustainability issues in Australia and has been the focus of a considerable amount of research and extension. The major land degradation issues include:

- Soil erosion
- Soil salinisation and sodicity
- Soil acidification
- Soil organic matter decline
- Soil nutrient decline/depletion
- Soil structure decline (compaction etc.).
- Provision or restriction of ecosystem services (such as maintenance of biodiversity, and carbon sequestration).

Attempts have been made to group the impacts of from land occupation and transformation into the following categories; impacts on biodiversity, impacts on biotic production potential and ecological soil quality (Mila i Canals et al. 2007). However, quantification of the environmental impacts of land occupation has rarely been attempted due to its complexity and data requirements. Indicators are difficult to define, particularly for broader environmental services. However some studies have described methods that are applicable to particular situations and more recently characterisation factors for land use (land occupation and land transformation) have been developed under a UNEP/SETAC Life Cycle Initiative (Koellner et al. 2012).

No studies were found in the literature that investigated the impacts of beef production on land occupation specifically, though Peters et al. (2011) did report nutrient flows and soil acidification at the farm level. Hence, a new set of relevant indicators were determined for the current study.

3.3.3 GHG Emissions

Agricultural sources contributed 14.6% of Australia's total GHG emissions in 2010 (DCCEE 2012). Of this, enteric methane was the largest contributor (67.8% of agricultural emissions). Three industries are the principal contributors to national enteric emissions (dairy cattle, sheep and beef cattle) and of these, beef cattle are by far the largest contributor because of the relative size of the beef herd. Beef production has a number of potential sources of GHG emissions in addition to enteric methane that also need to be accounted for. Emissions also arise from manure, fossil fuel energy use, and from emissions generated in the production of purchased inputs (such as fertiliser or grain). Emissions and carbon sequestration may also arise from land use change because of changes in vegetation and soil carbon levels, though there is a large degree of uncertainty surrounding the magnitude of these impacts and the methods that should be used when assessing these impacts.

3.3.3.1 Enteric Methane Processes

Enteric methane is the largest source of GHG across the life cycle of beef production (Cederberg et al. 2009a, Peters et al. 2010a, Verge et al. 2008). Consequently, this emission source has received the bulk of research to date into emission quantification and mitigation strategies. Enteric methane literature was reviewed recently by Cottle et al. (2011), and selected material is supplied here for context.

Enteric methane is produced in the digestive tract of ruminant livestock by microorganisms during anaerobic fermentation of the soluble and structural carbohydrates contained in the diet. The rate of enteric methane generation is influenced by the nutritional management of livestock and reflects the quality and balance of nutrients, energy and protein in the diet. Methane emissions from ruminant livestock typically represent a loss of 6-10% of gross energy intake (Johnson et al. 2003) and may be higher for cattle fed on tropical pastures common to the northern beef industry (Kurihara et al. 1999).

These losses represent a significant inefficiency in the digestive process, and reductions to methane emissions would improve feed energy use and the energy efficiency of the system.

A wide range of methods for reducing enteric methane emissions have been identified and reviewed by Cottle et al. (2011) and many others. These options fall broadly into three categories: i) rumen manipulation/alteration of rumen ecology, ii) breeding of 'low methane' animals, and iii) animal production management (herd reproduction, grazing management). These were reviewed in detail by Cottle et al. (2011).

A range of studies was reviewed to provide context to the enteric methane emissions estimated in this study. These are summarised, with relevant details, in Table 5.

TABLE 5 – ENTERIC METHANE EMISSIONS FROM BREEDING COWS AS PRESENTED IN THE LITERATURE

animal type	Live weight	Nutrition	Methane emission as reported (g/d)	Reported or Calculated annual methane emission (kg/hd/yr.)	Reference
cow	580-600	Best grazing management – rotational grazing + supplementation	-	67.5	(DeRamus et al. 2003)
cow	580-600	continuous grazing - some restricted access and weight loss	-	86.0	(DeRamus et al. 2003)
cow	506.2	rotationally grazed - lucerne	246	89.7	(McCaughey et al. 1999)
cow	516.2	rotationally grazed - grass	270	98.6	(McCaughey et al. 1999)

Emissions per animal unit show a degree of variability in the literature, largely due to differences in nutrition, genotype and feed additives known to reduce methanogenesis in the rumen.

It should be noted that on an animal basis, some counterproductive measures may also lead to reduced enteric methane production. For example, Kurihara et al. (1999) found that Brahman heifers fed on low quality Angleton grass produced less enteric methane per MJ of energy intake compared to a higher digestibility grass or grain. However, these cattle lost a considerable amount of weight on this diet compared to the other diets fed. Cottle & Nolan (2009) note that methane emissions could be reduced by selecting for cattle that have a lower feed intake and smaller mature weight, though this would also be counter to beef production goals.

However, beneficial findings have also been identified. Johnson and Johnson (1995) note that as dry matter intake increases, the proportional loss of gross energy intake to methane is reduced. Additionally these authors note that as digestibility and energy density increases, relative methane production declines. Consequently, pasture fed cattle supplemented with grain have been shown to produce less enteric methane as a proportion of gross energy intake (DeRamus et al. 2003). Likewise, cattle fed a highly digestible grass diet were found to produce lower emissions than those fed on low quality forage (DeRamus et al. 2003). Cattle fed on grain diets commonly produce less methane proportional to GE intake (Johnson & Johnson 1995).

While absolute methane emissions per animal (per day or per year) are useful for context, the focus of LCA research is the estimation of emission intensity relative to production, i.e. kg of methane per kg of beef. Increasingly this is being recognised by GHG researchers as a significant distinction when considering enteric methane emissions. This leads to a greater emphasis and interest in methane relative to intake (i.e. as a % of GE or DE) and the performance of the animals under investigation. For breeding animals, the number of calves produced and the live weight at weaning are the primary determinants of productivity, and have a very large impact on whole herd enteric methane efficiency.

Secondly, the average daily gain (ADG) of the young cattle post weaning is an important measure of efficiency. Where data are available for daily methane emissions and growth rate, the efficiency of production (kg of methane /kg of gain) can be determined.

Improvements that may be made in emission efficiency by manipulating herd production parameters have been investigated under Australian conditions by Hunter & Niethe (2009), Charmley et al. (2008) and McCrabb and Hunter (1999). These studies have identified improvements in GHG efficiency by improving weaning rate in the breeding herd and live weight gain in slaughter cattle. Overall estimated improvements were in the order of 30-55% reduction of methane per kilogram of beef produced.

However, the full implications of these improvements are yet to be considered. For example, the authors note that associated GHG emissions arising from the production of supplements or higher quality pastures have not been considered. These issues will be addressed by the LCA project.

3.3.3.2 Life cycle GHG emissions from beef production

A literature review was conducted across beef LCA studies in Australia and internationally to provide context for the current research. The review identified 17 LCA studies of beef production, 11 of which were sufficiently detailed to warrant inclusion in the review.

Most studies reported data on the basis of live weight or carcass weight, though few included post farm gate processing. Functional units, allocation procedures, global warming potentials and results were standardised using data from within the studies or through contact with the authors where possible.

Beef production from dairy systems was found to be quite different to ‘purpose grown’ beef production in several studies. Beef from dairy calves reduced emissions considerably (Nguyen et al. 2010, Williams et al. 2006), mainly because 85-92% of the emissions are typically allocated to milk production (Basset-Mens 2008). Studies where a proportion of the beef is derived from dairy calves are noted in Table 6 and are discussed in the following sections.

The rate of inclusion of beef from dairy sources is an important distinction between studies. European studies are particularly likely to include beef from dairy systems, because this contributes some 50% of total European beef production (Cederberg & Stadig 2003).

Enteric methane was consistently reported as the largest single emission source where data were disaggregated. The contribution from enteric methane was in the order of 50 – 76% in seven studies (Beauchemin et al. 2010, Casey & Holden 2006, Cederberg et al. 2009a, Cederberg et al. 2009b, Nguyen et al. 2010, Ogino et al. 2004, Verge et al. 2008). Contributions from enteric methane were highest from the Brazilian study (Cederberg et al. 2009a), where livestock production is based on pasture systems with low inputs from grain, fertiliser or other high energy inputs, and relatively low productivity (national average weaning rate of 54%, finishing age of slaughter cattle was reported as 4 years at 200 kg CW). Intensive production systems such as those practised in the northern hemisphere (i.e. Nguyen et al. 2010) resulted in lower relative contributions from methane because: i) rapid growth rate of slaughter cattle will result in lower methane emissions associated with livestock maintenance and therefore lower emissions per kg of beef produced, and ii) contributions from other sources such as carbon dioxide (related to fossil fuel usage) and nitrous oxide (related to the use of nitrogen fertilisers on pastures or crops) are generally higher with more intensive modes of production.

The second largest source of total GHG was from nitrous oxide (all sources combined), which contributed in the order of 20-35% for the four studies where these results were disaggregated (Beauchemin et al. 2010, Cederberg et al. 2009a, Cederberg et al. 2009b, Verge et al. 2008). One study (Edwards-Jones et al. 2009) included an organic case study which reported extremely high levels of nitrous oxide emissions (contributing more than 50% of overall emissions), which skewed the results from this study.

The remaining emissions from beef arise from CO_2 associated with fossil fuel usage throughout the supply chain (i.e. transport, farming operations and emissions embedded with products such as fertiliser). The contribution from this source, where results were disaggregated, ranged from as low as 2% for the Brazilian study (Cederberg et al. 2009a) to around 10% for a Canadian study (Verge et al. 2008).

Life cycle assessment links productivity and environmental performance. Hence, assessments are sensitive to biological productivity measures, particularly those related to breeding efficiency and feed conversion ratio (FCR). In general, higher productive efficiency leads to lower GHG. Improved feed efficiency reduces embedded emissions associated with grain usage, contributing to reduced GHG per kg meat. Reducing feed requirements will also decrease the throughput of nitrogen in the system, decreasing manure nitrous oxide emissions.

Improved breeding efficiency (i.e. higher weaning percentages, lower mortality rates to slaughter, shorter breeding intervals) will result in higher meat production from the breeding herd, and improved whole of system feed efficiency. This is particularly important for ruminants, because of the low number of progeny per breeder and high animal related emissions for the breeding herd. Several research projects have shown that higher productivity, even where this requires more intensive production, will lead to lower overall GHG. For example, Pelletier et al. (2010) and Peters et al. (2010a) both showed that grain finishing beef resulted in lower GHG than pasture finishing when all emission sources were accounted for. Improvements in productive efficiency were cited by four studies as a reason why meat production is becoming more efficient with respect to total GHG over time (Cederberg et al. 2009b, Verge et al. 2009, Verge et al. 2008).

Sensitive factors associated with feed production include the use of nitrogen fertiliser (which has a high level of embedded emissions) and the emissions of nitrous oxide, which are related to the total nitrogen cycling within the system. Systems that utilise leguminous pastures and crops should in principle result in lower GHG because of the reduced emissions associated with nitrogen fertiliser. However, these systems still generate nitrous oxide (if the IPCC methodology is followed) because of residual N added to the system (De Klein et al. 2006). Improvements would be observed for all livestock species where feed produced with low nitrous oxide emissions could be utilised. This is an advantage for a nation such as Australia, where the prevalence of dryland agriculture and relatively low annual rainfall in the cropping zones (typically less than 750mm average annual rainfall) leads to very low nitrous oxide emissions from cropping (tier 2 EF = 0.003 – DCCEE 2010) versus a default value of 0.01 for many European countries – IPCC 2006).

TABLE 6 – TOTAL GHG FROM BEEF LCA STUDIES REPORTED IN THE LITERATURE

Reference	Country	Data source	Production System	kg CO_2-e / kg LW [a]
Beauchemin et al. (2010)	Canada	Simulated farm study	Beef herd, calves weaned into feedlot from weaning. Feedlot duration is 11 months; weight at slaughter is 605 kg.	13.8
Cederberg et al. (2009b)	Sweden	National Inventory	Mixed national herd- 65% of beef from dairy industry.	10.9
Cederberg et al. (2009a)	Brazil	National Inventory	Specialist beef, pasture based system with low production and long finishing phase (inc. meat processing)	15.4
Casey & Holden (2006)[b]	Ireland	Farm Data	Conventional – specialist beef.	13
			Agri-environmental scheme – specialist beef.	12.2
			Organic – specialist beef.	11.1
Edwards-Jones et al. (2009)	Wales	Farm Data	Conventional specialist beef production.	16.2
			Organic specialist beef, pasture + hay and concentrates.	48.6
Nguyen et al. (2010)	Europe	Simulated farm study	Dairy calves finished at 12 months (weight at slaughter is 450 kg), fed on silage/grain diet.	8.8
			Beef herd, steers finished at 16 mts (weight at slaughter is 600kg). Semi-extensive pasture, hay and concentrate feeding system.	15.0
Ogino et al. (2004)	Japan	Simulated farm study	Japanese intensive production, imported feed, fully housed livestock. Slaughter at 28 mts, weight at slaughter is 722 kg.	15.1
Pelletier et al. (2010)	USA	Simulated farm study	Beef herd, slaughter cattle finished in feedlot from weaning. Feedlot duration is 10 months, weight at slaughter is 637 kg.	14.8
			Beef herd, slaughter cattle backgrounded on forage / hay then finished in feedlot. Feedlot duration is 5 months, weight at slaughter is 637 kg	16.2
			Beef herd, slaughter cattle finished on pasture for 15 months, slaughter wt, 505 kg.	19.2
Peters et al. (2010a)	Australia (VIC - 2004)	Farm Data	Organic specialist beef production (inc. meat processing)	9.6
	Australia (NSW 2002/2004)		Specialist beef, pasture/feedlot finishing (inc. meat processing).	8- 8.2
Verge et al. (2008)	Canada	National Inventory	Pasture/feedlot – 10% emissions reduction attributed to dairy calves in supply chain.	10.9
Williams et al. (2006) [b]	UK	National Inventory	Mixed national herd – beef from beef and dairy calves.	8.7
			Single enterprise beef production.	13.9

[a] For comparison between studies, data have been re-analysed to present data on a live weight basis. Wherever possible the assumptions presented in the original study were followed. In lieu of these data being available, a dressing percentage of 55% was used to back calculate live weight from (unallocated) carcass weight values. GWP were standardised to 25 for methane and 298 for nitrous oxide.

[b] These studies did not provide sufficient data to revise and standardise the GWP values.

4 Methodology

4.1 Goal Definition

The goal of the project was to investigate resource use and environmental impacts from two northern beef supply chains producing cattle for either export or domestic markets. A series of scenarios were modelled to investigate potential mitigation strategies with a particular focus on greenhouse gas mitigation.

4.2 Project Scope

4.2.1 Functional Unit

The functional unit represents the primary output from the supply chain and is closely related to the system boundary. Previous MLA research projects have used the functional unit '1 kilogram of Hot Standard Carcass Weight – HSCW', which is useful because it is widely used in the industry. However, HSCW does not align well with the system boundary, because it is measured part way through the slaughtering process. This unit is typically considered a primary output from farm production, but this is complicated by a number of co-products that need to be accounted for during meat processing (i.e. hides, edible offal etc.). A more straight forward approach is to use live weight as the main functional unit prior to slaughter, then to use a processed unit that reflects the whole processing stage (such as boned beef). For the purposes of the study, we applied a number of different functional units that align with different points in the supply chain. These are grouped under the following headings 'farm gate', 'processor gate' and 'consumer'.

Farm Gate

Breeding systems (cow-calf) were compared using a 360 kg LW steer after breeding and backgrounding, but prior to finishing. This weight of animal was selected as a typical 'feeder' steer, suitable for entering several different finishing systems. The functional unit was 1 kg of beef produced on-farm (Live weight - LW). Total farm production was compared per kilogram of beef produced at the farm gate (LW).

Processor Gate

Results post-processing are presented per kilogram of beef at the processor gate (boned beef).

Consumer

Results are presented per kilogram of beef consumed in the home (either in Australia or Japan).

4.2.2 System Boundary

The system was divided into a foreground and a background component. Foreground system data were collected from the cattle production supply chain, from breeding to slaughter (identified by the system within the dashed box – Figure 6). For both supply chains, self-replacing breeding herds were used, with all impacts associated with breeding replacement bulls and heifers accounted for. Major components of the system are shown in Figure 6 (SW supply chain). This does not imply that other components were excluded. The NE system boundary is similar to

Figure 6, with the omission of the feedlot component. Processes included within the dashed line represent the foreground system.

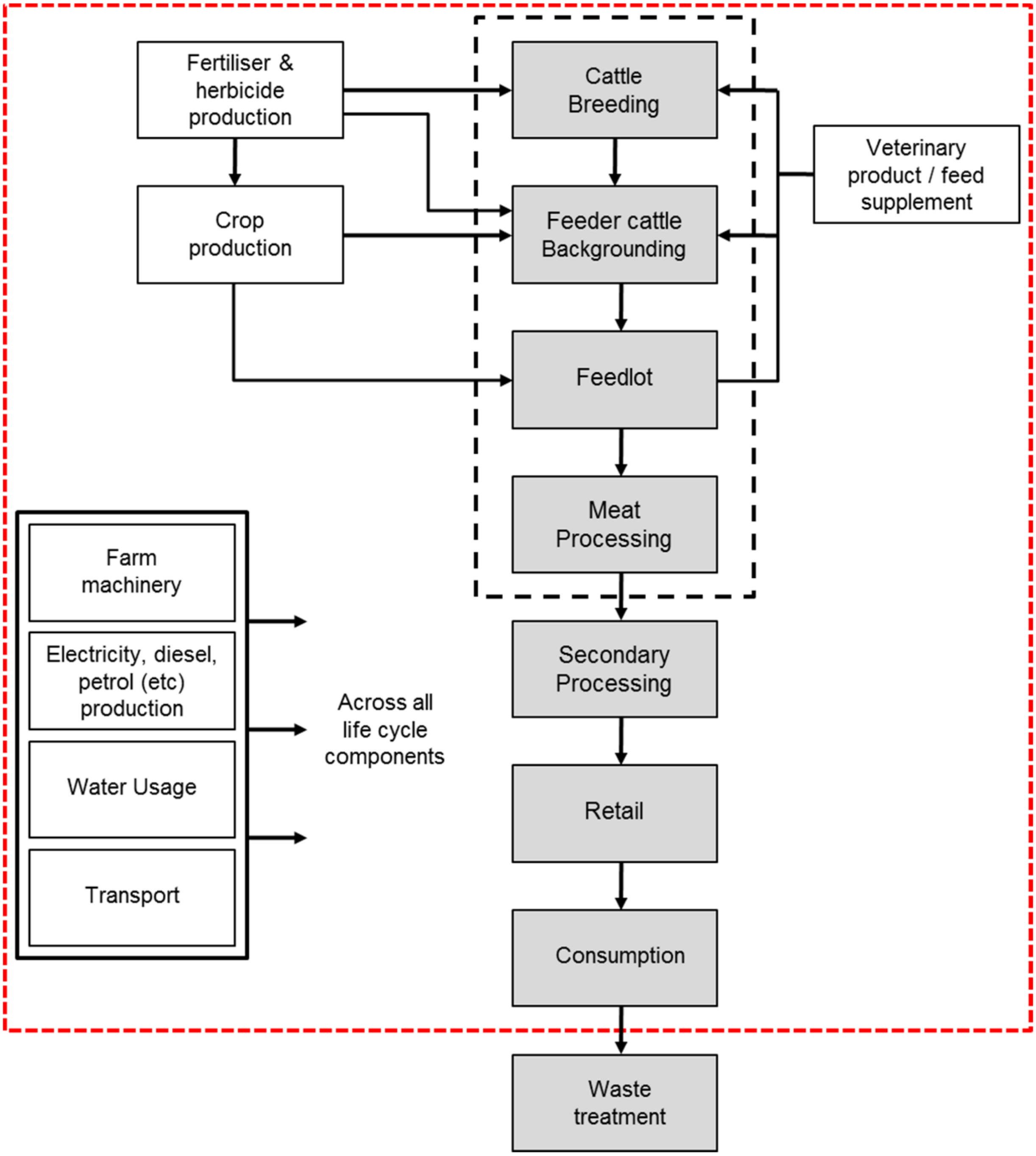

FIGURE 6 – SYSTEM BOUNDARY FOR THE SW SUPPLY CHAIN

The post processing supply chain was investigated in detail, and a sub-system boundary for the domestic supply chain is shown in Figure 7.

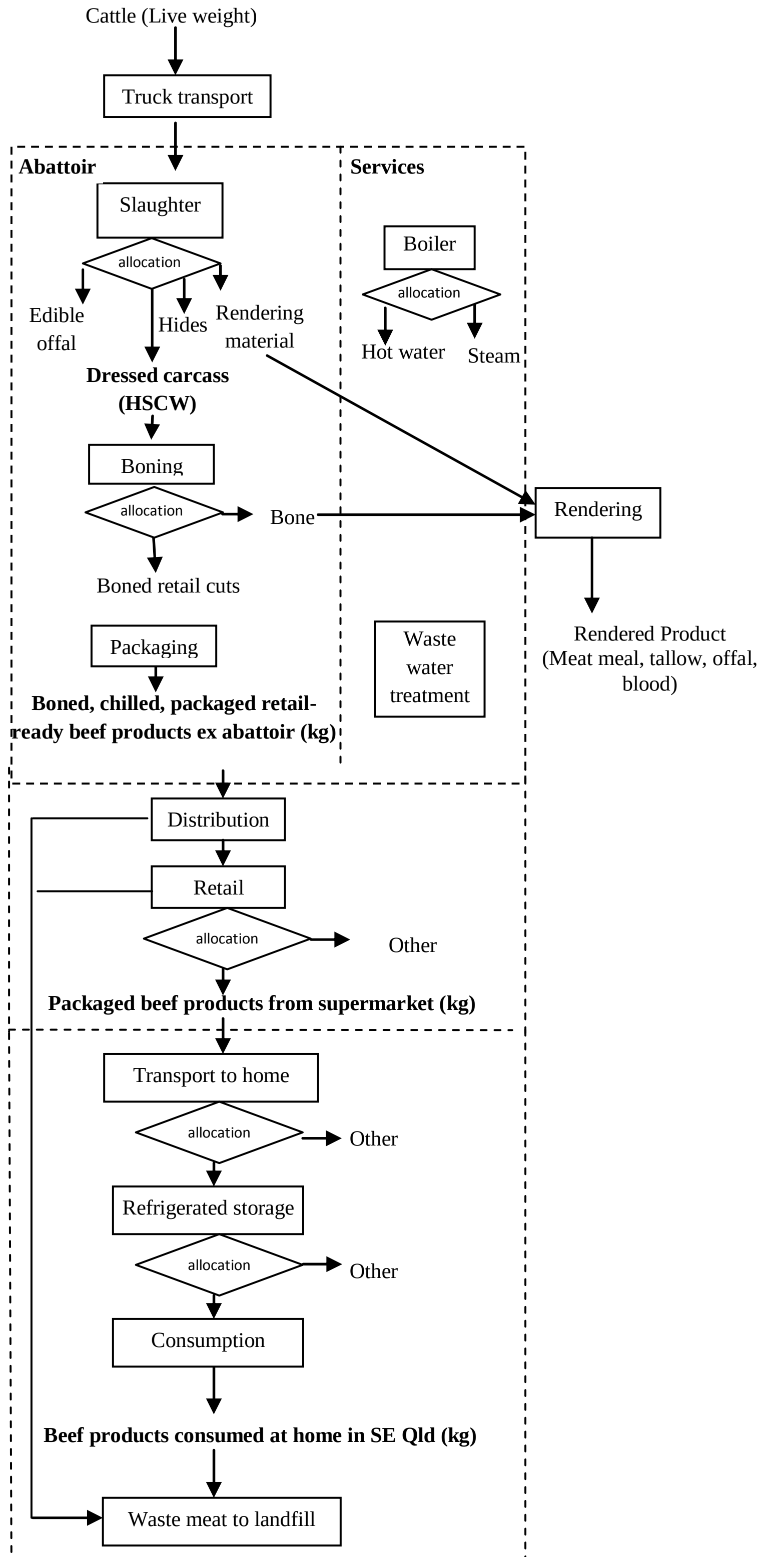

FIGURE 7 – SYSTEM BOUNDARY FOR THE MEAT PROCESSING AND POST PROCESSING STAGES OF THE SW SUPPLY CHAIN (NOTE: ALLOCATION TO 'OTHER' REFERS TO OTHER GROCERY PRODUCTS AT THE RETAIL, TRANSPORT AND STORAGE STAGES)

4.3 Resource use and Environmental Impact Categories

4.3.1 Energy Demand

Energy demand was assessed using the Cumulative Energy Demand (CED) indicator (Frischknecht et al. 2007), measured in mega joules (MJ) using Lower Heating Values (LHV). Cumulative energy demand includes energy from non-renewable and renewable sources, but excludes energy contained in plants that is digested by animals.

4.3.2 Water Use

The water use inventory was developed using the Consumptive Fresh Water use (consumptive water use) indicator. Additionally, the impact assessment method ‘stress weighted water use’ was used (Pfister et al. 2009). A detailed explanation of the inventory methods and data are provided in Appendix 3 – Water Use Inventory.

TABLE 7 – WATER USE CLASSIFICATIONS AND METHODS

Water use reporting category	Units	Description	Noted exclusions
Consumptive Fresh Water Use (synonymous with blue water)	L	All consumptive water uses throughout the supply chain.	Flows of water through treatment systems that are then released for use in the environment or other systems. The criteria in this case were that the water must be beneficially utilised in replacement of other fresh water sources.
Stress weighted water use	L H_2O-e	All consumptive water uses multiplied by the relevant WSI value, summed across the supply chain and divided by the global average WSI (after Ridoutt et al. 2011a).	Exclusions noted above for consumptive water use

4.3.3 Land Occupation

Land occupation has not previously been included in most Australian agricultural LCAs. Land occupation is a standard category within LCA and is a simple aggregation of the land area required to produce a given product. We have included land occupation (measured in m^2 yr.) with two land occupation classifications i) use of non-arable pasture land, ii) use of arable land for cultivation or pasture. A detailed explanation of the inventory methods and data are provided in Appendix 2 – Land Occupation and Nutrients.

4.3.4 Grain use and Human edible protein and energy conversion ratio (HP-FCE, HE-FCE)

Grain use, and more specifically ‘human edible energy and protein’ were identified as resource inputs using a detailed inventory of grain use throughout the supply chain. Grains were characterised to determine the human edible protein (kg) and energy content (MJ/kg), taking into account milling losses where relevant.

4.3.5 Land Occupation Impacts

We chose Soil Depletion Potential (potential soil loss via water erosion) as the land occupation impact indicator in the current study. A detailed explanation of the inventory methods and data are provided in Appendix 2 – Land Occupation and Nutrients.

4.3.6 Eutrophication Potential

While nutrient loss to waterways is a topic of national concern in Australia, there was insufficient primary research available to develop regionalised characterisation factors and quantify eutrophication for the supply chains investigated in this study. Eutrophication Potential was qualitatively assessed for the grazing farms using a risk assessment tool developed for Australian farms (the Farm Nutrient Loss Index, or FNLI – Melland et al. 2007). A detailed explanation of this method is provided in Appendix 2 – Land Occupation and Nutrients.

4.3.7 Greenhouse Gas Emissions

Greenhouse gas emissions were determined from all livestock (enteric methane, manure emissions) and from purchased inputs (energy, feed, fertiliser etc) throughout the supply chain. Emission estimates were based on recent Australian research and the Australian National Greenhouse Gas Inventory (NGGI) (DCCEE 2010). The study applied updated GWPs (see Table 8). Potential emissions from land use change were not included in the study.

TABLE 8 – THE GLOBAL WARMING POTENTIAL OF MAJOR GREENHOUSE GASES

Greenhouse Gas	Kyoto compliant 100 yr. GWPs (1990 baseline) applied by the Australian National Inventory (DCCEE 2010)	100 year GWPs – IPCC (2007) [a]
Carbon Dioxide	1	1
Methane	21	25
Nitrous Oxide	310	298

[a] Solomon *et al.* (2007).

4.4 Inventory Development

The goals of the project required collection of detailed data from two production systems utilising two alternate finishing systems, grain and grass. The project used a case study approach, with supply chains to be selected that were broadly representative of production systems. However, the results were not intended to be representative of ‘Queensland beef’.

All primary data were sourced from commercial businesses. To address variability in production, foreground data were collected for a minimum of two years (farm inputs) and for a minimum of two years for livestock production and herd parameters.

4.4.1 Collection of Foreground Data

Site visits were carried out throughout the supply chains to collect foreground data and conduct a broad assessment of biophysical characteristics on each farm. The main data sources were:

- Farm financial accounts (covering purchased inputs and livestock sales).
- Production records (covering livestock production and movements on the farm).
- A farm survey of natural resource management practices and natural resource condition (providing more detailed information on soils, vegetation, water, erosion and nutrient management).

Energy demand was determined from purchased energy (electricity, diesel, petrol) and transport records for purchased inputs used by the farm. Inventory data are presented in Appendix 1 – Farm and Feedlot Inventory Data.

4.4.2 Modelling of Foreground Processes

Where data were not available for some inputs and outputs in the foreground system these were modelled or estimated from literature values. Key modelled inputs include water use and feed intake (dry matter intake). These data were modelled from climate data, herd characteristics and performance. Similarly, important outputs such as enteric methane emissions could not be measured, but were modelled based on the livestock herd. Other data such as nutrient losses were estimated from a theoretical mass balance model using parameters sourced from the literature.

4.4.3 Background Data

Background data for upstream processes such as generation and supply of energy and purchased products such as fertiliser were sourced from the Australian LCI database (Life Cycle Strategies 2007). Energy demand associated with the manufacture of purchased inputs such as fertiliser was based on either the Australian LCI database (Life Cycle Strategies 2007) where available, or the European EcoInvent (2.0) database (Frischknecht et al. 2005). Some processes (i.e. feed grain production) were sourced from data collected by FSA Consulting.

4.5 Selection of Supply Chains

4.5.1 Supply Chain Characteristics

The project investigated two case study supply chains for the state of Queensland, Australia. Queensland is the largest beef production region in Australia, but has a wide variety of production systems and levels of productivity throughout the state (Bortolussi et al. 2005). The scope of the study as determined by MLA was to investigate one system producing grass fed, heavy export bullocks (Japan ox) and another system producing domestic grain-finished cattle. Hence, the study was focused on comparison of market types and the systems used to produce these market types.

Grass fed export bullocks for the Japanese market are subject to market criteria for weight, age and fat cover. Liveweight at slaughter is 570-660 kg (which is close to mature weight for most Queensland bullocks), and age is typically 3 to 5 years old based on dentition (6-8 tooth). Younger cattle may be sold as Japan ox if higher growth rates can be achieved, but these cattle will often be preferentially sold into higher value markets. The Japan ox market is most suitable for older slaughter cattle produced at lower growth rates. For example, the lifetime growth rate required to achieve 600 kg LW at 4 years is 0.4 kg/d. Because of the lower liveweight gain requirements, Japan ox is a preferred market for lower rainfall areas.

Cattle for the domestic market in Australia are predominantly produced in rangeland grazing areas and finished on grass, forage or grain. Where cattle are finished on grain, the feeding period is a relatively short interval (50-70 days) immediately prior to slaughter. Cattle for the domestic market are slaughtered at younger ages (generally under 2 years of age, based on ossification score) at a liveweight of 430-460 kg. To achieve this weight at less than two years of age the lifetime growth rate must be in the order of 0.6 kg/d. Growth rates during grain finishing of around 1.7 kg/d are typical, resulting in a total live weight gain during grain finishing of 20-25% of slaughter liveweight. Feedlots are mainly located in the southern part of Queensland and feeder cattle are sourced from right across the state, and from southern states, depending on availability and price.

4.5.2 Selection Criteria

The supply chains were selected to be broadly representative of the type of systems used to produce cattle for these two markets. However, considering the size of the state, it was not possible to represent all geographical regions and no attempt was made to achieve this. Instead, a case study approach was used, and the representativeness of the results is discussed later in the report. Participant farms were selected according to the availability of data and the degree to which the production system was consistent with common practice in the region and state.

The grass fed bullock supply chain was located in the northern part of the state, west of Townsville. This supply chain is broadly representative of the region, where many cattle are produced for the Japan ox market. A general description of the supply chain is shown in Figure 8. This supply chain is abbreviated in the report as the NE (north east) supply chain.

The domestic, grain-finished supply chain was located in the south-western part of the state, and consisted of a breeding / backgrounding property, a feedlot and a meat processing plant. Exact locations of the farms are not provided to protect the confidentiality of the industry participants. A general description of the supply chain is shown in Figure 8. This supply chain is abbreviated in the report as the SW (south west) supply chain.

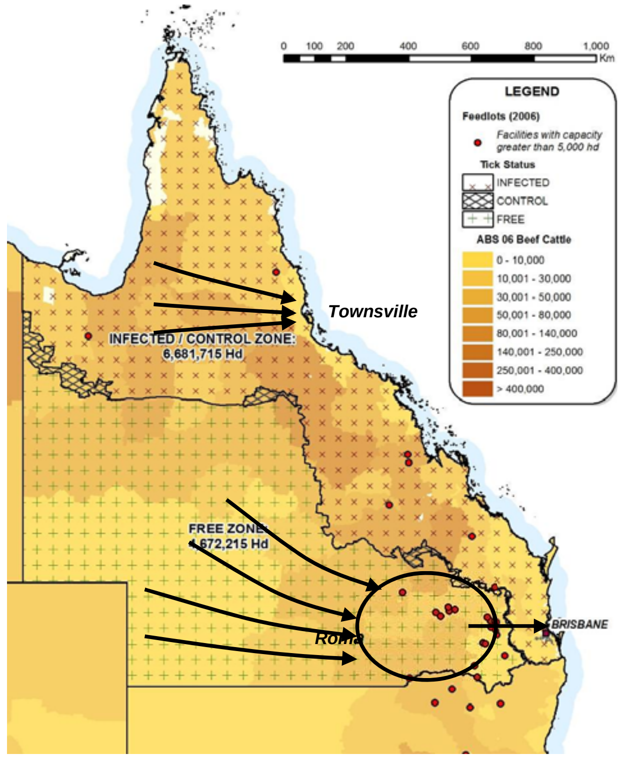

FIGURE 8 – BEEF CATTLE DISTRIBUTION IN QLD (ABS 2006) INCLUDING LOCATION OF MAJOR FEEDLOTS (CIRCLED) AND THE TICK LINE

4.6 Case Study Farms

4.6.1 NE QLD Supply Chain

The property has an annual average rainfall of 642 mm, with a high degree of inter-annual variation. Soils on the property are predominantly low fertility yellow earths, with a smaller proportion (approximately 25%) of more fertile brown cracking clays which are located along three main waterways on the property. Vegetation on the property consists of open eucalypt forest (yellow earths) and blackwood/gidgee scrub (clay soils). Grasses consist of perennial tussocks, with *Bothriochloa* spp., *Dicanthium* spp., *Heteropogon contortus* and *Themeda triandra* being the main species. Some areas of the property have been cleared by chaining the standing woody vegetation. Over some cleared areas this has been combined with sowing introduced pastures (Buffel grass – *C. ciliarus* and *Stylosanthes spp.*). Additionally, these introduced species have been sown into areas of open forest, particularly on the clay soils.

Approximately 1200 cows are mated each year in a self-replacing herd. Bullocks are grown out to slaughter at approximately three years of age and 600 kg LW. Most heifers are kept on-farm and joined to calf at three years of age. Heifers and cows are pregnancy tested and all empty animals are culled. Characteristics of the breeding herd are provided in Table 9.

TABLE 9 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE NE SUPPLY CHAIN

Breeder cattle		
Production parameter	Units	Average
Weaning rate	%	61.9
Breeder culling rate	%	25.0
Herd bulls	%	10.0
Mortality rate	%	2.0
Weaning weight (6 months)	kg LW	179
Backgrounding		
ADG (birth to 360 kg LW)	kg /d	0.61
Age at 360 kg	mths	17.6

The weaning rate for this farm was relatively low compared to southern Queensland beef production, but was similar to the average of 54 north Queensland beef producers (64.2%) collated by Bortolussi et al. (2005). Likewise, weaning weights were similar to the regional average (170kg) reported by Bortolussi et al. (2005) for north Queensland.

The herd was divided into three sub-systems: breeding, backgrounding and finishing. These sub-systems and livestock movements are shown in Figure 9. Where possible, impacts were directly attributed to the system that generated the impacts (i.e. enteric methane, drinking water use etc.). However, whole farm inputs such as fuel use could not be readily allocated to one system over another. Therefore, these inputs were divided based on the proportion of dry matter consumed by the livestock in each sub-system.

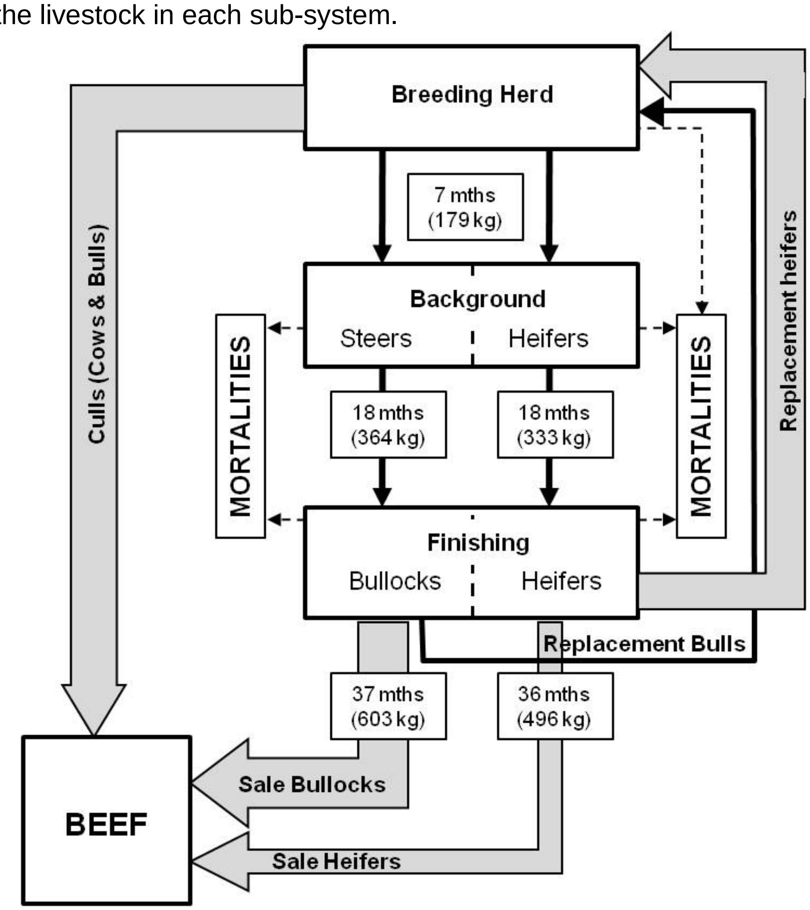

FIGURE 9 – LIVESTOCK SUB-SYSTEMS AND CATTLE MOVEMENTS IN NE SUPPLY CHAIN

4.6.2 SW QLD Supply Chain

Breeding and Backgrounding

The south west Queensland breeding and backgrounding property was located north of Mitchell, Queensland. The property has an average annual rainfall of 568 mm. Soils on the property range from brown to grey clays, typically associated with brigalow (*Acacia harpophylla*) and belah (*Casuarina spp.*). The property has two distinct land types; i) open forest with native pastures, and ii) cleared pasture land, sown with Buffel. The breeding herd is predominantly grazed in the forested part of the property while the improved pastures are used for backgrounding slaughter cattle prior to lot feeding.

Approximately 650 cows are mated each year in a self-replacing herd. Steers and sale heifers are backgrounded on farm to 340-350 kg (~17mths) prior to transfer to the feedlot. Replacement heifers are kept on-farm and joined to calve at three years of age. Characteristics of the breeding herd are provided in Table 9.

TABLE 10 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE SW SUPPLY CHAIN

Breeder cattle		
Production parameter	Units	Average
Weaning rate	%	79.1
Breeder culling rate	%	16.2
Herd bulls	%	8.1
Mortality rate	%	2.5
Weaning weight (7 months)	kg LW	221
Backgrounding		
ADG (birth to 360 kg LW)	kg / d	0.64
Age at 360 kg	mths	17.0

The performance of the SW breeding property was similar to the survey average of 46 beef producers in the Maranoa South West collated by Bortolussi et al. (2005). Bortolussi et al. (2005) reported average weaning rates of 77.1% and weaning weights of 216 kg LW.

The herd was divided into two sub-systems, breeding and backgrounding, and impacts were assigned in the same way described for the NE supply chain.

Feedlot

Foreground data were collected from production and accounting records over a two year period, and included feed commodities, energy usage, total water use and cattle movements. Detailed cattle productivity data (i.e. average daily gain, feed intake) and accurate cattle movements (head days) were available from herd management software used by the feedlot. Herd productivity data (Table 11) and feed data were used to calculate manure production, emissions and enteric methane production. A modified version of BEEFBAL (QPIF 2004) (an Excel spreadsheet mass balance model for feedlots) was used for this task. Greenhouse gas emissions from manure management relied on these estimations. A detailed explanation of feedlot modelling methods can be found in the appendices.

TABLE 11 – PRODUCTION CHARACTERISTICS FOR THE FEEDLOT

Production parameter	Units	Feedlot
Entry weight	kg LW	360
Days on feed	days	54
ADG	kg / d	1.7
Exit weight	kg LW	437
Mortality rate	%	0.6%
Intake	kg DM / d	8.9
FCR		5.3

4.6.3 Alternative Backgrounding-Finishing Scenarios

A series of scenarios were investigated to explore options to reduce greenhouse gas 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. The scenarios were as follows:

- Fast growth rate backgrounding (forage crops and supplementation) and feedlot finishing
- Fast growth rate backgrounding and finishing on forage crops with supplementation.
- Fast growth rate backgrounding and finishing on leucaena.
- 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.

For the SW property, a scenario was also modelled where steers were grown out to ~600 kg LW for comparison. Production data and assumptions are provided in Table 12 and Table 14.

TABLE 12 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR ALTERNATIVE BACKGROUNDING-FINISHING SCENARIOS (NE)

	Growth rate		Age at slaughter (mths)	Slaughter weight (kg LW)
Finishing system	Backgrounding-finishing phase – kg/d	Birth to slaughter - kg/d		
Leucaena	0.7	0.71	26.6	603
Leucaena with supplement	1	0.89	21.2	603

TABLE 13 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR ALTERNATIVE BACKGROUNDING-FINISHING SCENARIOS (SW)

	Growth rate			Age at slaughter (mths)	Slaughter weight (kg LW)
Finishing system	Backgrounding phase – kg/d	Finishing phase – kg/d	Birth to slaughter - kg/d		
Leucaena and Forage finishing	0.7	0.8	0.80	16.8	440
Leucaena and Feedlot finishing	0.7	1.66	0.90	15.3	437

TABLE 14 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR ALTERNATIVE MARKETS AND FINISHING SYSTEMS – EXPORT STEERS

	Growth rate			Age at slaughter (mths)	Slaughter weight (kg LW)
Finishing system	Backgrounding phase – kg/d	Finishing phase – kg/d	Birth to slaughter – kg/d		
Forage/suppl. and feedlot finishing (75d)	1.2	1.7	0.83	22.8	608
Forage/suppl. and feedlot finishing (120d)	1.2	1.70	0.88	23.5	654
Leucaena, FL finishing	1.1	1.7	1.10	18.5	651
Grass, Forage and supplementary feed	0.8	0.9	0.71	27.7	630
Leucaena with revegetation	0.75	0.75	0.68	28.9	630
Irrigated Leucaena and supplementary feed	1.1	1.1	0.79	25.1	630
Low input grass finishing	0.455	0.35	0.47	39.4	598

These scenarios were modelled to target different market types with different slaughter weights. 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 Beuchemin 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%.

The Leucaena scenarios assumed 11% lower enteric methane after Kennedy & 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, we assumed ½ ha. of land was available for regrowth. Regrowth was assumed to be Acacia (Brigalow) open forest, with carbon storage of 32 t / ha. (after Fensham & Guymer 2009). Carbon sequestration was annualised over 100 years to provide an annual rate of 0.32 t C ha. yr.

TABLE 15 – FORAGE GRAIN SUPPLEMENT RATION

Commodities (protein content in brackets)	Units	Amount
Sorghum (10%)	kg	250.0
Wheat (13%)	kg	405.0
White fluffy cottonseed	kg	200.0
Wheat Straw	kg	80.0
Recycled oil	kg	25.0
Feed additives	kg	40.0
Total	kg	1000.0

4.7 Meat Processing

Meat processing plants use large quantities of energy and clean water in the slaughtering process. Energy is used for machine operation and cooling of the slaughtered carcass. Water is used to maintain high food hygiene standards. Water is used for watering and washing livestock, cleaning process equipment and work areas, and washing carcasses. Cleaning makes up a large proportion (around 50%) of water use.

Foreground data were collected from the meat processing plant where cattle from the SW supply chain are processed. The processing plant was located in SE Queensland. Foreground data included water and energy use, waste stream processes and production data. The plant included rendering facilities, which were included within the system boundary.

Cattle from the NE supply chain were slaughtered in Townsville. Data were not obtainable from the Townsville processing plant, and were obtained from an alternative meat processing plant in South East Queensland which slaughter a similar type of cattle. Data collection covered energy and water use, waste treatment and production. The plant included rendering facilities, which were included within the system boundary.

4.8 Handling Co-Production

Co-products were identified at four points in the foreground system. The grazing farm produces both prime beef and beef from cull breeders. At the feedlot, both beef (live weight) and manure are produced. Manure is sold as a fertiliser replacement and soil conditioner and has a very low value compared to beef. Meat processing results in the production of several co-products.

Co-production of beef from cull cows and beef from prime cattle was handled using a mass allocation process at the point of slaughter. This results in equal burdens being attributed to the cull and prime beef. The allocation choice was made reflecting the underlying function of both meat products. Beef from cull cows enters the manufacturing beef market and a small proportion of cuts (rump, sirloin etc.) are retailed at a discount price. Prime beef is sold primarily onto the fresh beef market, with a smaller proportion of off-cuts entering the manufacturing beef market. The primary function of all these products is the provision of a high protein food source for human consumption. Nutritionally, there is little difference between mince that originally came from a cull cow, and sirloin steak from a prime animal, hence the burdens were considered equivalent. At the feedlot, co-production of manure and beef was handled by system expansion. Using this process, the avoided mass of fertiliser replaced by feedlot manure was subtracted from the system using the method described by Wiedemann et al. (2010a).

The third allocation point was at the point of slaughter. The primary product at this point is beef (HSCW) and the secondary products are hides (leather), offal (edible for human consumption and pet food), blood and bone (rendered into blood meal, meat meal and meat and bone meal) and fat (tallow).

The fourth allocation point was during boning, where the primary product is meat (primal or retail cuts) and the secondary products are primarily bone and fat, which are rendered to produce meat meal and meat and bone meal.

The primary method of handling co-production throughout the supply chain was using system expansion. However, to investigate the sensitivity of this process and enable comparison with other studies, economic and mass allocations were also applied. The processes and details involved in the handling of co-production are described in Table 16.

Co-products were identified at four points in the foreground system. Decisions regarding co-production are described in Table 16.

TABLE 16 – METHODS FOR HANDLING CO-PRODUCTION

Stage in Supply Chain	Product and co-product (in brackets)	Method	Reason for choosing method for handling co-production
Grazing farm	Sale cattle (cull cows)	No allocation applied	There was no clear rationale for discriminating between beef from prime and cull cattle, considering the end product from both classes of cattle (beef) is suitable for human consumption. Functional differences relate to markets and consumer preferences but not nutritional quality. The output from all systems was taken to be total beef produced from all classes of saleable cattle.
Feedlot	Beef live weight (nutrients contained in manure).	System expansion	Where nutrients were used to replace synthetic fertilisers, a system expansion process was used.
Meat Processing	Carcase (hot standard carcase weight – HSCW) and slaughter by-products.	System expansion. Economic and mass allocation applied for comparison.	A hybrid mass allocation and system expansion approach was taken. Impacts were allocated on a mass basis between the carcase mass, edible offal and hides. The system was expanded to replace the mass of protein and energy produced in the minor slaughter by-products (e.g. tallow) based on the most common use for these products. Mass allocation and economic allocation methods were applied for comparison and results are reported in the discussion. Substitution products and allocation factors are reported in Table 17 and Table 18.
Meat Processing	Retail meat (bone-in) and secondary products (pet food, rendering products).	System expansion. Economic and mass allocation applied for comparison.	The system was expanded to replace the mass of protein and energy produced in the slaughter by-products based on the most common use for these products. Mass allocation and economic allocation methods were applied for comparison and results are reported in the discussion. Substitution products and allocation factors are reported in Table 19 and Table 20.

In Australia, 'Hot Standard Carcase Weight' is a clearly defined industry classification for the sale of cattle. Australian cattle are generally marketed on the mass of HSCW and weighed at the meat processing plant soon after slaughter. From a producer point of view, HSCW is the output of the primary production system and they are paid only for carcase mass. However, there are a number of co-products from the slaughter of cattle (as with other species) that have value and must be taken into account in the LCA.

At the point of slaughter, co-production was handled using a hybrid approach. Impacts were allocated on a mass basis between the carcase mass, edible offal (for human consumption) and hides. The justification for this approach is provided as follows: carcase mass and edible offal were considered functionally equivalent, because both have similar nutritional properties for human consumption. As the primary function of livestock products is human consumption to meet nutritional requirements (primarily as a protein source) there was no clear justification to differentiate between these products using an allocation process. Allocating between meat and hides was similarly done on a mass basis as hides are a very important primary product for a diverse range of high value products (shoes, furniture, car upholstery etc.). Leather from cattle hides cannot be considered a by-product in the market. At certain points in history animals have been slaughtered primarily for hides not meat. Hence, we felt that allocating on a mass basis was reasonable.

A number of minor by-products are generated from meat processing and the associated rendering process. These products (blood/bone/meat meal, pet food and tallow) were handled using system expansion on mass basis with products of similar energy and protein content. Meal products are typically used as feed sources for poultry and pigs. They were therefore substituted for soybean meal on a 'protein equivalent' basis. Likewise, pet food was substituted for soybean meal on a protein equivalent basis. Tallow was similarly substituted for canola oil.

Specific assumptions regarding the handling of co-products from slaughter are provided in Table 17 and Table 18 for the NE QLD and SW QLD supply chains respectively.

TABLE 17 – MEAT PROCESSING FACTORS (HSCW) FOR NE QLD SUPPLY CHAIN

Slaughter Products	Mass of product (kg)	Mass Allocation Factors	Economic Allocation Factors	System expansion substitution products
Hot carcase weight	1 000	71.8%	88.6%	-
Edible offal	76	5.4%	3.3%	Considered to be functionally equivalent to carcase weight
Secondary Products				
Hides	122	8.8%	4.8%	-
Blood meal	10	0.7%	0.2%	Soymeal and sorghum– on protein and energy equiv. basis
Meat and bone meal	83	6.0%	1.2%	Soymeal and sorghum– on protein and energy equiv. basis
Tallow	89	6.4%	1.6%	Canola oil
Pet food	12	0.9%	0.2%	Soymeal and sorghum– on protein and energy equiv. basis
Totals	1393	100%	100%	

TABLE 18 – MEAT PROCESSING FACTORS (HSCW) FOR SW QLD SUPPLY CHAIN

Slaughter Products	Mass of product (kg)	Mass Allocation Factors	Economic Allocation Factors	System expansion substitution products
Hot carcase weight	1 000	68.0%	84.4%	-
Edible offal	86	5.9%	3.6%	Considered to be functionally equivalent to carcase weight
Secondary Products				
Hides	162	11.0%	8.5%	
Blood meal	10	0.7%	0.2%	Soymeal and sorghum– on protein and energy equiv. basis
Meat and bone meal	84	5.7%	1.1%	Soymeal and sorghum– on protein and energy equiv. basis
Tallow	104	7.1%	1.8%	Canola oil
Pet food	23	1.5%	0.4%	Soymeal and sorghum– on protein and energy equiv. basis
Totals	1470	100%	100%	

While HSCW is a standard classification in the beef industry, it does not align well with stages in the supply chain. In literal terms, hot carcases are weighed soon after slaughter, part way through meat processing. A useful 'end point' in meat processing is a product that can be sold to wholesalers or retailers. In many instances this is a chilled, trimmed carcase, or primals (wholesale meat portions). A further processing stage is required to trim and slice the product into retail portions. Assumptions relating to further processing of hot carcases to chilled, boned retail meat are shown in Table 20 for the NE QLD and SW QLD supply chains respectively.

TABLE 19 – MEAT PROCESSING FACTORS (RETAIL MEAT) FOR NE SUPPLY CHAIN

Slaughter Products	Mass of product (kg)	Mass Allocation Factors	Economic Allocation Factors	System expansion substitution products
Bone-in retail beef	1000	84.6%	97.8%	-
Secondary Products				
Meat and bone meal	88	7.5%	0.9%	Soymeal and sorghum– on protein and energy equiv. basis
Tallow	94	8.0%	1.3%	Canola oil
Totals	1183	100%	100%	

TABLE 20 – MEAT PROCESSING FACTORS (RETAIL MEAT) FOR SW SUPPLY CHAIN

Slaughter Products	Mass of product (kg)	Mass Allocation Factors	Economic Allocation Factors	System expansion substitution products
Bone-in retail beef	1000	84.0%	97.7%	-
Secondary Products				
Meat and bone meal	85	7.2%	0.9%	Soymeal and sorghum– on protein and energy equiv. basis
Tallow	106	8.9%	1.4%	Canola oil
Totals	1191	100%	100%	

4.9 Modelling and Uncertainty

All modelling was done using SimaPro ™ version 7.3. This included a sensitivity analysis of model parameters and an uncertainty analysis. Uncertainty within the model relates to both natural variability in inventory data and uncertainty related to assumptions made during the modelling process. The uncertainty analysis was based on data ranges determined during the inventory phase. Uncertainty was assessed using a Monte Carlo analysis in SimaPro™. Monte Carlo analysis is a means of handling cumulative uncertainty within the system. Rather than estimating a theoretical minimum and maximum (i.e. the cumulative lowest and cumulative highest values), the analysis develops a distribution pattern from 1000 randomly selected scenarios, based on the possible range of values for each parameter. These results are used to provide the 95% confidence interval for the results.

5 Results – Farm Gate

The 'farm gate' results section is divided into three sections. Results from the two breeding farms (cow-calf production) are presented 'per kilogram of live weight' using a standardised 220 kg weaner. The second section compares the backgrounding and finishing phases for each market category. Results in this section are reported 'per kilogram of live weight at the farm gate, immediately prior to slaughter'. These represent the main results for the cattle produced on each farm.

The last section provides scenarios that model GHG mitigation approaches, with results presented 'per kilogram of live weight at the farm gate, immediately prior to slaughter'.

5.1 Breeding

5.1.1 Resource Use

The resource use assessment covered energy, water, land and grain use. At the breeding farm we did not assess the ratio of human edible protein and energy consumed and produced, because this indicator is more suited to classes of livestock ready for meat processing (see Section 6.2.1).

Energy use was significantly higher for the NE compared to the SW farm (see

Table **21**). Energy use at both farms was mainly associated with direct farm energy use (diesel, petrol and electricity, contributing 50-55% for the NE and SW farms respectively) and purchased farm inputs and services (supplementary feed, telecommunications, financial, repairs etc.).

Consumptive water was higher at the NE farm compared to the SW farm (see

Table 21). Consumptive water use was governed by drinking water requirements (a function of climate and herd productivity) and water losses from the drinking water supply system. Neither farm used irrigation. Drinking water use varied from 170 L/kg weaner LW (NE) to 130 L/kg weaner LW (SW), with differences being primarily driven by herd efficiency and climate, both of which favoured the SW property. Where drinking water was sourced from a creek (consumed directly) or a bore (via a tank and trough system) the losses were negligible, and consumptive water use was equivalent to drinking water. However, evaporative losses were considerable from farm dams (137 and 107 L/kg weaner for the NE and SW farms respectively). Dams contributed 42% (NE) and 58% (SW) of drinking water, but evaporation rates were lower at the SW farm, corresponding to a lower contribution from dam evaporation at this farm than may have been expected.

Stress weighted water use (see

Table **21**) was lower than consumptive water use at both farms, though this difference was much greater at the SW farm (where the WSI was lower). Stress weighted water use is a measure of the impact of using water, which is proportional to the degree of water stress in a given region. As both farms were located in regions with comparatively low levels of water stress, the stress weighted water use was much lower than consumptive water use, indicating that the impact of using this water was relatively low.

TABLE 21 – ENERGY AND WATER USE FOR WEANERS PRODUCED FROM TWO QLD BEEF CATTLE FARMS

Farm	Energy Demand	Consumptive Water Use	Stress weighted water use
	MJ / LW	L / kg LW	L H_2O-e / kg LW
NE QLD weaners (averaged at 220kg LW)	4.7 ± 11%	308.8 ± 36%	56.9 ± 36%
SW QLD weaners (averaged at 220kg LW)	2.5 ± 8%	237.0 ± 41%	8.5 ± 40%

The vast majority of the land occupied for production is classified as non-arable rangelands grazing native pastures, with minimal disturbance of native vegetation. Land occupation (non-arable) was 948 m^2 and 1621 m^2/kg weaner LW for the NE and SW farms respectively, and arable land use (associated with supplementary feed use) was 0.4m^2 and 0.2 m^2 for the NE and SW farms respectively.

5.1.2 Environmental Impacts

Impacts on land (Soil Depletion Potential, Soil Carbon Flux Potential) and climate (Greenhouse Gas Emissions) were assessed. A qualitative assessment of eutrophication is also included for the two breeding properties.

Soil Depletion Potential (soil erosion) varied considerably between the two farms, mainly in response to differences in soil type and susceptibility to erosion. These values are subject to a high degree of uncertainty however. Soil depletion is a concerning environmental impact because of the severe consequences of excessive soil loss and the long time frame required for soil formation and this may require further investigation.

TABLE 22 – SOIL DEPLETION POTENTIAL FOR WEANER PRODUCTION FROM TWO QLD BEEF CATTLE FARMS

Farm	Soil depletion Potential
	kg
NE QLD weaners (averaged at 220kg LW)	55.6 ± 43%
SW QLD weaners (averaged at 220kg LW)	2.0 ± 45%

The assessment of greenhouse gases took into account direct emissions (livestock, land and energy related emissions) and indirect emissions or sequestration sources from land occupation. Greenhouse gas emissions were driven by enteric methane (87-88%), with smaller contributions from direct and indirect manure emissions (9% for both farms). Emissions from pasture forage and grain production were low (1%), as were emissions from farm services (2-3%). The contribution to greenhouse gas emissions (not including soil carbon flux) is shown in Figure 10.

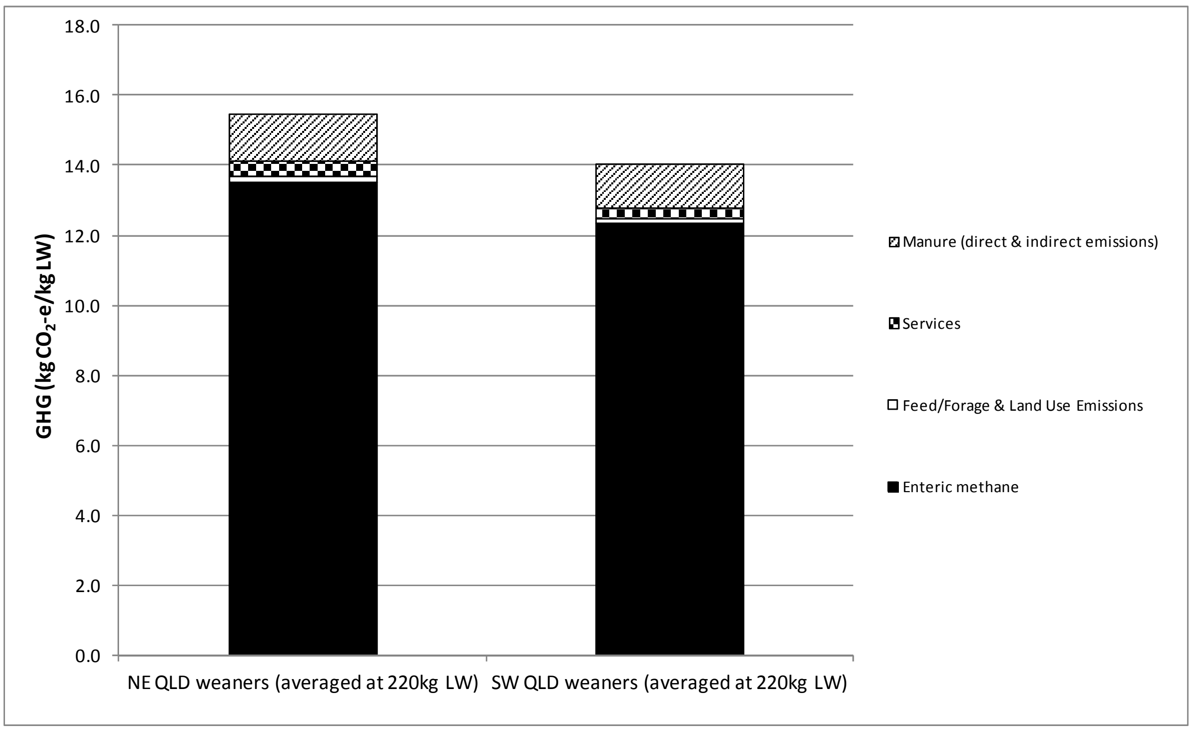

FIGURE 10 – GREENHOUSE GAS EMISSIONS FOR WEANERS PRODUCED FROM TWO QLD BEEF CATTLE FARMS

Considering the pronounced relationship between herd productivity and environmental impacts such as GHG (Hunter & Niethe 2009), particular notice was taken of this effect. Herd efficiency can be measured in kilograms of beef per kilogram of dry matter intake (whole herd). This is governed primarily by weaning percentage, mortality rate and average daily gain in the young cattle and replacement cow herd. Weaning rate varied from 61.9-79.1%, while mortalities were relatively low on both farms (2-2.5%). Growth rates for the calves (to weaning) varied from 0.5 – 0.9 kg/d to 220 kg LW. The lower productivity from the NE farm corresponded with 10% higher GHG emissions.

Soil Carbon Flux Potential is a measure of the change in soil carbon for soils throughout the supply chain. Values were very low (positive) for both farms (0.007-0.008 kg CO_2-e/kg LW). Positive values indicate potential emissions of carbon from soils, and these losses were associated with the use of feed supplements rather than changes on the grazing farms (where soil carbon was assumed to be in equilibrium). This assessment was subject to a high degree of uncertainty (see Table 23). Carbon losses (positive values) arose from cultivated land occupation and these were predicted based on Dalal & Chan (2001).

TABLE 23 – SOIL AND GHG EMISSIONS FOR STEER AND HEIFER PRODUCTION FROM TWO QLD BEEF CATTLE FARMS

Farm	Soil Carbon Flux Potential	GHG emissions
	kg CO_2-e	kg CO_2-e
NE QLD weaners (averaged at 220kg LW)	0.007 ± 80%	15.4 ± 21%
SW QLD weaners (averaged at 220kg LW)	0.008 ± 83%	14.0 ± 20%

On both farms the risk of phosphorus and nitrogen loss in runoff was rated as low, primarily because of the low nutrient application rates, low stocking rates, high levels of ground cover (generally >85%) and predominantly perennial pastures. Similarly, N and P loss via subsurface lateral flow was rated low to medium. The medium rating on the NE farm appeared to be an anomaly, but may be understood by taking into account that the model assumes the lowest input of phosphorus fertiliser is 0-11kg P ha.yr, while at the farm phosphorus had never been used.

Nutrient losses via deep drainage were rated as low for both farms. The low risk ratings relate to the summer dominant rainfall pattern and relatively low annual rainfall (<650mm annually). This, together with the negligible nutrient application rates and the predominance of perennial pastures resulted in low expected nutrient losses.

TABLE 24– NUTRIENT LOSS RISK ASSESSMENT FROM THE NE AND SW GRAZING PROPERTIES

	NE QLD		SW QLD	
Nutrient loss Pathways	Risk Rating	Score (max 8)	Risk Rating	Score (max 8)
Phosphorus				
Runoff	Low	3	Low	2
Subsurface lateral flow	Medium	4	Low	2
Deep drainage	Low	3	Low	2
Nitrogen				
Runoff	Low	3	Low	2
Subsurface lateral flow	Low	4	Low	3
Deep drainage	Low	3	Low	1
Gaseous emissions	Low	1	Low	1

5.2 Backgrounding and Finishing

The two farms grew cattle out to different markets with different systems. We divided the post-weaning stage into backgrounding (220-360kg LW, steers and heifers, both farms) and finishing. At the NE farm, steers were grown out from 360 to ~600kg LW (5 year av. 603 kg) on grass with some supplementary feed for the Japan Ox market. Heifers were grown out and joined to calve at three years. Heifers failing to calve (and older cows failing to calve) were sold for slaughter. At the SW farm, steers and heifers (excluding replacements) were grain finished for 54 days.

5.2.1 Resource Use

Energy demand was similar between the farms (4.7 and 4.3 MJ/kg LW) for the NE and SW farms respectively. The similar energy demand was driven by different factors at the two farms. The NE farm used relatively higher amounts of diesel, much of which was associated with clearing woody regrowth. Energy demand at the SW supply chain was very low for the breeding and backgrounding, and a large proportion of energy demand was associated with grain feeding. Energy demand with grain feeding was related to upstream grain production, feed milling and transport.

Consumptive water use was influenced most by the characteristics of the specific farm. Water use was higher at the NE farm because of the higher drinking water requirements and evaporation rates at this farm, combined with lower herd productivity and lower growth rates, though this was partly offset by higher slaughter weights.

Stress weighted water use was much lower than consumptive water use for both farms (45.9 and 7.7 L H_2O-e/kg LW for the NE and SW supply chains). The lower water stress values when compared to consumptive use reflect the impact that water use has on the stress on water resources in the region. The NE farm was located in a region with slightly higher water stress, resulting in higher stress weighted water volumes. Despite this, the impact of using water was considered to be low compared to the global average water stress.

Arable land occupation showed a contrast between the two farms. The NE supply chain used very small areas of arable land (0.5m^2/kg LW) while the SW supply chain used considerably more (3.9 m^2/kg LW), which was all associated with grain production for the lot-feeding phase. Non-arable land use was higher for the SW supply chain, which was driven by the large area of land used for grazing the breeding herd. Low levels of grazing intensity may appear to be a less efficient use of resources, but may have better sustainability outcomes because land use impacts may also be lower.

TABLE 25 – ENERGY AND WATER USE FOR BEEF PRODUCTION FROM TWO QUEENSLAND SUPPLY CHAINS

Farm	Energy Demand			Consumptive Water Use			Stress weighted water use		
	MJ / kg LW			L / kg LW			L H_2O-e / kg LW		
NE Japan Ox (603 kg LW)	4.7	±	8%	247.9	±	35%	45.9	±	34%
SW Domestic Market (Grain finished) 435 kg LW	4.3	±	5%	183.4	±	35%	7.7	±	29%

TABLE 26 – LAND AND GRAIN USE FOR BEEF PRODUCTION FROM TWO QUEENSLAND SUPPLY CHAINS

Farm	Occupation of arable land			Occupation of pasture land (non arable)			Grain use		
	m^2 / kg LW			m^2 / kg LW			kg / kg LW		
NE Japan Ox (603 kg LW)	0.5	±	4%	655.6	±	14%	0.1	±	4%
SW Domestic Market (Grain finished) 435 kg LW	3.9	±	14%	1032.8	±	16%	0.8	±	6%

5.2.2 Environmental Impacts

Soil Depletion Potential was 39.4 and 5.0 kg / kg LW for the NE and SW farms respectively. However, the estimates are subject to a high degree of uncertainty because of the scale of the erosion assessment.

TABLE 27 – ENVIRONMENTAL IMPACTS FOR BEEF (LW) PRODUCTION FROM TWO QUEENSLAND SUPPLY CHAINS

Farm	Soil depletion Potential			Soil Carbon Flux Potential			GHG emissions		
	kg			kg CO_2-e			kg CO_2-e		
NE Japan Ox (603 kg LW)	39.4	±	42%	0.011	±	82%	12.9	±	15%
SW Domestic Market (Grain finished) 435 kg LW	5.0	±	41%	0.119	±	83%	11.2	±	16%

Greenhouse gas emissions were lowest from the SW (11.2 kg CO_2-e /kg LW) compared to the NE (12.9 kg CO_2-e / kg LW) – see Figure 11. The largest single contributor to GHG was enteric methane, which ranged from 87% (NE) to 85% (SW). Contributions from manure emissions were very similar (9% - NE and 10% - SW), as were contributions from feed production (ration commodities, feed supplements or pasture) which were 2% and 3% for the NE and SW supply chains respectively. Interestingly, contribution from feed production for the grain fed cattle did not greatly contribute to emissions, because of the short feeding period used by this supply chain (54 days). Emissions at the NE farm arose from supplement use and directly from pastures.

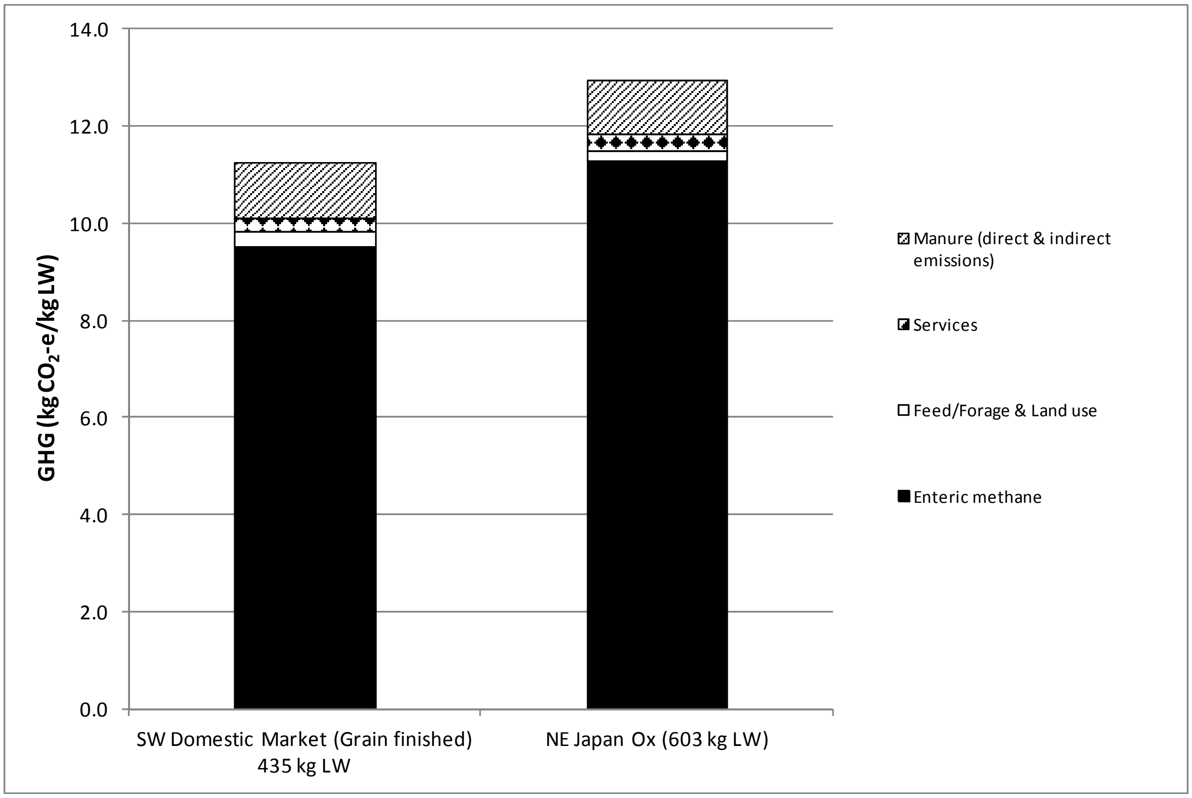

FIGURE 11 – GREENHOUSE GAS EMISSIONS PER KILOGRAM OF LIVE WEIGHT AT THE FARM GATE FOR THE TWO QLD SUPPLY CHAINS

5.3 GHG Mitigation Scenarios

Two sets of mitigation scenarios were modelled. The first set of scenarios focused on options that were reasonably achievable on each farm without major changes to the type of cattle/markets and were possible utilising existing land resources. The second scenarios explored options that included a broader range of backgrounding/finishing possibilities available in Queensland. These options were modelled for one supply chain only (the SW supply chain) but were generally applicable to both, though some of these would involve transporting the NE cattle to more productive backgrounding and lot feeding regions in central Queensland (~600 km south).

5.3.1 On-farm GHG Mitigation Scenarios

The main options investigated focused on productivity improvements (whole herd feed efficiency) to achieve reductions in enteric methane intensity (CH_4/kg LW). The main opportunities to achieve this were to improve weaning rates and growth rates for slaughter cattle. Numerous approaches have been suggested to improve the efficiency of Queensland beef production. Most are focused on more intensive management (i.e. pregnancy testing, controlled joining) or improved nutrition (supplementation, controlled/rotational grazing, seasonal feeding to improve nutrition for specific animal classes (such as cows on their second calf or breeders prior to joining, or early weaning). Improved nutrition may be achieved through greater inputs (i.e. supplements) a change in the pasture base (i.e. introduction of legumes), use of forage cropping or more intensive pasture management such as cell or rotational grazing. Such changes must be taken into account in modelling mitigations. We chose in this study to focus on improved weaning rates (NE farm) and growth rates in slaughter cattle. Additionally, we investigated scenarios that enabled carbon storage (sequestration) in either soil or vegetation.

We modelled improvements in animal performance that may be achieved by sowing legume pastures (in this case Leucaena). Leucaena is a popular perennial tree grown and managed to provide high value feed in conjunction with pastures (i.e. Buffel grass). The scenarios modelled were based on data reported by Radrazzini et al. (2011a, b) and Dalzell & Shelton (2007). Two scenarios were modelled. The first involved a strategic improvement in nutrition by sowing leucaena for the breeding herd. This was used to improve conception rates (particularly from cows on their second calf), and growth rates could also be slightly improved in the slaughter herd. The second scenario included the same improvements as the first scenario; however supplementation was also carried out. This resulted in higher growth rates than the first scenario for the slaughter herd. The scenarios were intended to be broadly applicable to farms in the regions where the case studies were carried out. Results are shown for these scenarios in Table 28.

TABLE 28 – NE QLD GHG MITIGATION FARM SCENARIOS

Emission/sequestration source	Leucaena – high wean (kg CO_2-e/kg LW)		Leucaena with supplementary feeding – high wean and high ADG (kg CO_2-e/kg LW)	
	no regrowth	**with revegetation**	**no regrowth**	**with revegetation**
Enteric methane	8.5	8.5	7.9	7.9
Feed/Forage production and land use	0.5	0.5	0.5	0.5
Services	0.3	0.3	0.2	0.2
Manure (direct & indirect emissions)	0.9	0.9	0.8	0.8
Sub-Total GHG	**10.2**	**10.2**	**9.5**	**9.5**
Soil carbon flux potential	-0.8	-0.8	-0.4	-0.4
Vegetation carbon flux potential - leucaena	-1.0	-1.0	-0.6	-0.6
Vegetation carbon flux potential - regrowth	0.0	-3.1	0.0	-3.6
Total Land Use Change emissions	**-1.8**	**-4.9**	**-1.0**	**-4.6**
Net GHG	**8.4**	**5.3**	**8.6**	**4.9**

Compared to standard production at the NE farm, emissions were 21-26% lower, as a result of the higher weaning rate, higher growth rate and slightly lower modelled enteric methane emissions from cattle grazing Leucaena. We modelled changes in soil carbon and vegetation in the Leucaena plantation, and modelled changes in carbon from revegetation of grazing lands made available after changing from pasture to Leucaena. This resulted in carbon sequestration (reported as negative CO_2-e emissions) of -1.0 to -4.9 kg CO_2-e/kg LW. Net emissions (livestock emissions net of sequestration) was 34-62% lower than emissions from the standard scenario.

For the SW supply chain, mitigations focussed on changes to the backgrounding and finishing systems. Two options were modelled; Leucaena for the backgrounding phase (220-360 kg LW) and finishing either on forage with supplementary grain feeding (4 kg/hd d), or lot feeding. Results are shown in Table 29.

TABLE 29 – SW QLD GHG MITIGATION FARM SCENARIOS

Emission/sequestration source	Leucaena Backgrounding, Forage Finishing (440 kg) (kg CO_2-e/kg LW)		Leucaena Backgrounding, Feedlot Finishing (435 kg) (kg CO_2-e/kg LW)	
	no regrowth	**with revegetation**	**no regrowth**	**with revegetation**
Enteric methane	9.3	9.3	8.8	8.8
Feed/Forage and land use	0.4	0.4	0.4	0.4
Services	0.2	0.2	0.3	0.3
Manure (direct & indirect emissions)	1.0	1.0	1.1	1.1
Sub-Total GHG	**10.9**	**10.9**	**10.6**	**10.6**
Soil carbon flux potential	-0.1	-0.1	-0.1	-0.1
Vegetation carbon flux potential - leucaena	-0.3	-0.3	-0.3	-0.3
Vegetation carbon flux potential - Regrowth	0.0	-0.9	0.0	-0.9
Total Land Use Change emissions	**-0.4**	**-1.3**	**-0.4**	**-1.3**
Net GHG	**10.5**	**9.6**	**10.2**	**9.2**

Compared to the standard production in the SW supply chain, the mitigations reduced emissions by 3% (Leucaena+forage) and 6% (Leucaena+feedlot). Emissions net of sequestration potential were 6-18% lower than standard production, with the largest offsets coming from the inclusion of areas of revegetation.

Mitigation potential for livestock emissions was greatest at the NE farm, partly because livestock emissions were higher to begin with from this farm, and partly because there was a longer growing/finishing period (with the heavier slaughter weight) over which to apply the mitigation strategy. We modelled a 'high weaning rate' scenario where we assumed weaning could be improved to 80%, which represents a practical maximum for this region, and is higher than is likely to be achieved in most seasons. Consequently, the mitigation potential achievable is expected to be slightly lower.

5.3.2 Industry GHG Mitigation Scenarios

The farm scenarios were modelled to broadly utilise the infrastructure and land types available, and produce cattle to similar market specifications. There is also potential for a broader number of mitigation scenarios that investigate alternative markets and finishing systems that may be achieved in Queensland. Production data for each scenario were provided in Table 14. While the scenarios were selected as GHG mitigations, we also investigated the impact on resource use (energy, water and cultivated land). This allowed consideration of trade-offs that may occur across the impact categories, rather than between GHGs only. Total GHG emissions (see Figure 12) were 17-31% lower for all scenarios compared to a standard grass finished Japan Ox type animal (~600 kg LW, 39 mths). The greatest GHG reductions (31% and 29%) came from the high growth rate irrigated Leucaena scenarios, either with or without feedlot finishing. The forage+feedlot scenarios (either 75 or 120d) resulted in a 22% and 25% reduction in GHG, while the grass, forage and grain supplementation scenario and the Leucaena with revegetation scenarios resulted in a 17% reduction in GHG. When the impacts of GHG fluxes from vegetation and soil were taken into account there were some different trends. The Leucaena scenarios were modelled with modest levels of soil carbon sequestration and also vegetation carbon sequestration in the Leucaena plantation. One Leucaena scenario was also modelled with revegetation, resulting in an additional source of carbon sequestration. In contrast, net GHG emissions tended to be slightly higher from the grain and forage feeding scenarios because some soil carbon losses were assumed from cultivation. When all these factors were accounted for, the net GHG emissions showed a slightly higher mitigation potential for the irrigated Leucaena scenarios (32% reduction in net GHG), while the mitigation potential was slightly lower from the supplementary grain feeding and feedlot scenarios (20-23%). The greatest contrast came with the Leucaena and revegetation scenario, where the mitigation potential was considerably greater (39% reduction) as a result of carbon storage in the revegetation area.

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 (5570 L / kg LW) and arable land (12 m^2 / kg LW). Irrigated Leucaena was selected because this system enables high annualised growth rates (1.1 kg/d) with an energy supplement such as molasses. This can also be achieved for a longer feeding period, making this a reasonable scenario for finishing cattle year round. Both grain finishing scenarios utilised a backgrounding stage with forage crops and supplementary grain feeding to achieve high growth rates (1.2 kg/d) before entering the feedlot. The feedlot scenarios utilised higher amounts of fossil fuel energy (for grain production and feedlot operations), higher amounts of grain and higher amounts of arable land.

The scenario utilising grass, forage and supplementary feed produced intermediate results. GHG mitigation potential was moderate, energy demand was similar to the feedlot scenarios, water use was similar to the other low impact scenarios as was arable land use, but grain use was lower, reflecting the greater use of forage crops rather than grain.

The Leucaena with revegetation scenario, in contrast to the others, utilised very little additional energy, water, arable land or grain. Mitigation potential was not as high as most other scenarios when considering only the reduction in emissions. However, when the sequestration potential was included the offset was more substantial. We assumed in this scenario that Leucaena was grown in a dryland situation on marginal land that could not be used for regular cultivation and was therefore classified as non-arable.

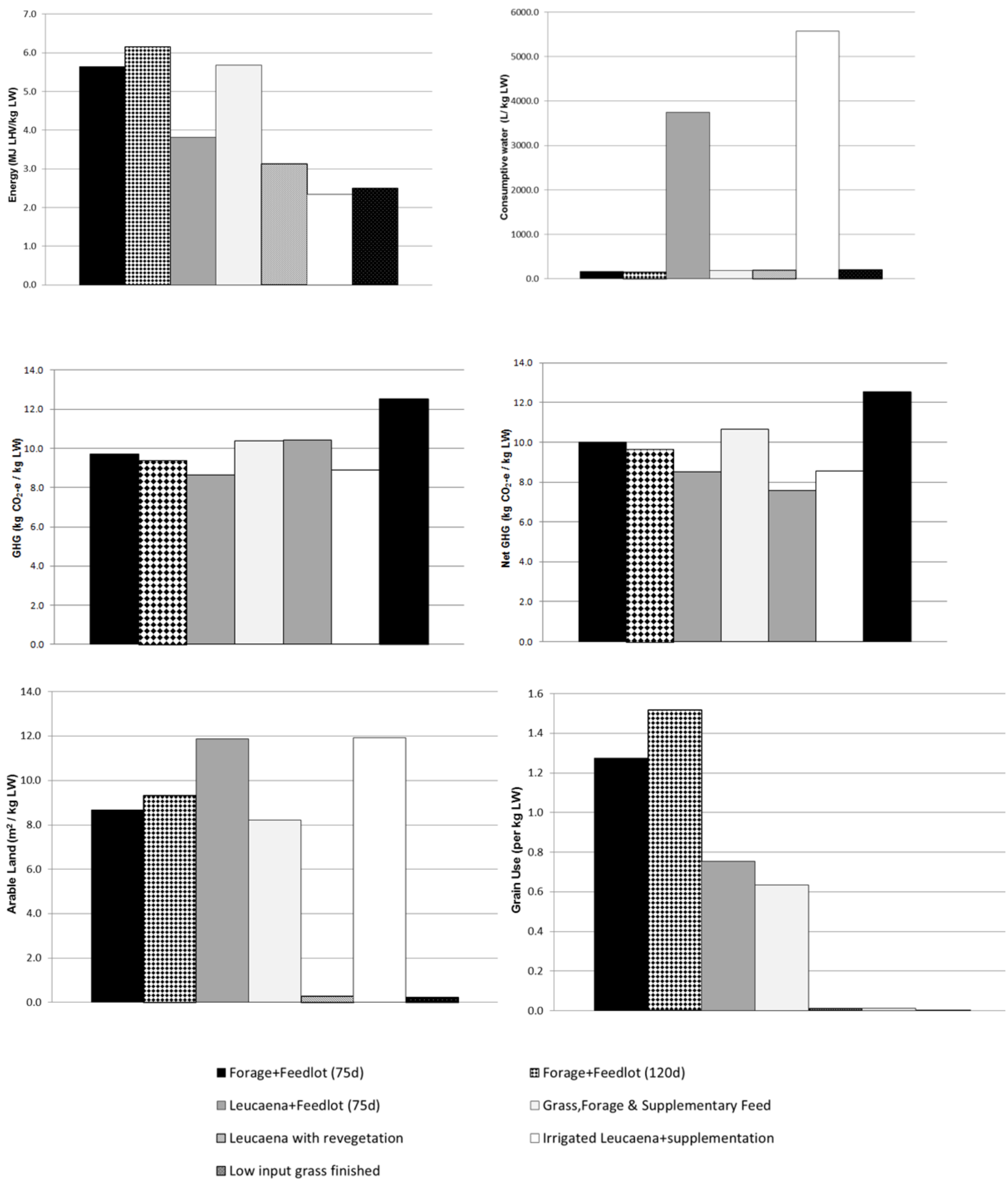

FIGURE 12 – MITIGATION SCENARIOS FOR BACKGROUNDING AND FINISHING STEERS SHOWING IMPACTS ON ENERGY DEMAND, STRESS WEIGHTED WATER USE, GHG, NET GHG, ARABLE LAND OCCUPATION AND GRAIN USE

TABLE 30 – RESOURCE USE FOR BEEF (LW) PRODUCTION FROM SEVEN ALTERNATIVE MARKETS/FINISHING SYSTEMS FOR SW QLD

Farm	Energy Demand	Consumptive Water Use	Stress weighted water use	Occupation of arable land	Occupation of pasture land (non arable)	Grain use	Human edible energy	Human edible protein
	MJ / LW	L / kg LW	L H_2O-e / kg LW	m^2 / kg LW	m^2 / kg LW	kg / kg LW	MJ / kg LW	kg / kg LW
Forage+Feedlot 75d	5.6 ± 9%	164.7 ± 31%	8.1 ± 23%	8.7 ± 14%	816.1 ± 16%	1.27 ± 8%	19.1 ± 8%	0.2 ± 8%
Forage+Feedlot 120d	6.2 ± 7%	148.4 ± 32%	7.8 ± 22%	9.3 ± 11%	767.3 ± 16%	1.52 ± 6%	22.8 ± 6%	0.2 ± 6%
Leucaena+Feedlot	3.8 ± 6%	3750.8 ± 32%	132.3 ± 22%	11.9 ± 6%	735.6 ± 16%	0.75 ± 6%	11.3 ± 6%	0.1 ± 6%
Grass,Forage & Supplementary Feed	5.7 ± 6%	190.0 ± 33%	9.0 ± 26%	8.2 ± 6%	807.8 ± 16%	0.63 ± 6%	9.5 ± 4%	0.1 ± 4%
Leucaena with revegetation	3.1 ± 7%	191.0 ± 31%	7.3 ± 29%	0.3 ± 4%	833.0 ± 16%	0.01 ± 11%	0.2 ± 9%	0.002 ± 7%
Irrigated Leucaena+supplementation	2.3 ± 7%	5570.0 ± 43%	195.0 ± 43%	11.9 ± 14%	756.0 ± 16%	0.01 ± 10%	0.2 ± 9%	0.001 ± 8%
Low input grass finished	2.5 ± 8%	199.0 ± 38%	7.2 ± 36%	0.2 ± 5%	962.0 ± 16%	0.004 ± 3%	0.1 ± 3%	0.001 ± 3%

TABLE 31 – ENVIRONMENTAL IMPACTS FOR BEEF (LW) PRODUCTION FROM SEVEN ALTERNATIVE MARKETS/FINISHING SYSTEMS FOR SW QLD

Farm	Soil depletion Potential	Soil Carbon Flux Potential	GHG emissions
	kg / kg LW	kg CO_2-e / kg LW	kg CO_2-e / kg LW
Forage+Feedlot 75d	4.3 ± 43%	0.3 ± 97%	9.7 ± 16%
Forage+Feedlot 120d	4.1 ± 39%	0.3 ± 80%	9.4 ± 14%
Leucaena+Feedlot	1.4 ± 38%	0.001 ± 107%	8.6 ± 15%
Grass,Forage & Supplementary Feed	5.7 ± 36%	0.3 ± 83%	10.4 ± 15%
Leucaena with revegetation	4.1 ± 45%	-0.4 ± -55%	10.4 ± 14%
Irrigated Leucaena+supplementation	1.2 ± 43%	-0.13 ± 56%	8.9 ± 16%
Low input grass finished	14.6 ± 45%	0.01 ± 83%	12.5 ± 14%

6 Results – Post-Processing

Results are presented in this section using two functional units that represent different points in the supply chain. The first of these is **one kilogram of boned beef ready for distribution to wholesale/retail.** This represents the main output from meat processing, where minimal further processing is required for retail. This is the first point at which meat from different species can be compared without major differences in product yield. Results for boned beef reflect the loss of mass from meat processing, consequently impacts increase proportionally. As discussed in Section 4.8, co-products were handled using a system expansion approach for minor by-products (meat meal, tallow, pet food etc.) and a mass allocation approach for meat and hides.

The second unit is **one kilogram of beef consumed in the home.** This functional unit includes transport, retail, packaging, home storage and cooking, either for meat purchased in Australia or Japan.

6.1 Boned Beef

6.1.1 Resource Use

Following assessment of meat processing, the magnitude of the impacts increased with a change of functional unit. The primary difference when comparing boned beef results with live weight is associated with the loss of product mass at the point of meat processing and processes used to account for co-products. It is also partly due to increased impacts directly associated with meat processing.

Energy use during meat processing was a major contribution to overall impacts, while water use was a relatively smaller contribution. Meat processing uses an appreciable amount of energy, primarily in the form of electricity, gas and/or coal for steam generation. Water use is not high in comparison to use on-farm.

The system expansion process used for handling minor co-products such as meat meal resulted in substantial ‘offsets’ for water, grain and arable land use in particular. Following the system expansion approach, the total burden of resources and impacts are attributed to the meat product. The system is ‘expanded’ to account for the impacts that would be required to produce *avoided* or substitution products that can replace the co-products in the market. The co-products handled in this way were meat meal, blood meal, tallow and pet food. The appropriate avoided (or substitution) products were soymeal and canola oil for the protein meals and tallow respectively. This reflects the market for these products reasonably well in Australia, where protein meals from beef production form valuable inputs for pig and poultry diets, offsetting the use of plant based proteins such as soymeal. The consequence of applying this approach was to provide ‘deductions’ or negative impacts to the system, corresponding to the avoided soy and canola production. In particular, there were substantial ‘deductions’ attributed for stress weighted water (because a proportion of the soymeal production was modelled from irrigated soymeal) and for grain use, arable land occupation and human edible protein and energy. In the case of arable land and grain use (Table 33) for the NE supply chain, these offsets were larger than the impacts from the supply chain, resulting in ‘negative’ impacts. These results can be interpreted as follows; the production of co-products from beef production offset the total use of arable land and stress water weighted use throughout the whole supply chain, resulting in ‘negative’ resource use. In effect, cattle production in the pastoral zones may reduce the pressure on arable land use, grain supply, human edible protein and energy and stress weighted water use through the production of these by-products.

TABLE 32 – ENERGY AND WATER USE FOR BONED BEEF PRODUCED FROM TWO QLD BEEF CATTLE FARMS

Supply chain	Energy Demand	Consumptive Water Use	Stress weighted water use
	MJ / kg boned beef	L / kg boned beef	L H_2O-e / kg boned beef
NE QLD Boned Beef	15.2 ± 6%	496.3 ± 37%	77.2 ± 44%
SW QLD Boned Beef	16.8 ± 4%	356.2 ± 38%	-4.5 ± 29%

TABLE 33 – ARABLE AND NON-ARABLE LAND OCCUPATION, GRAIN USE AND HUMAN EDIBLE PROTEIN AND ENERGY USE FOR BONED BEEF PRODUCED FROM TWO QLD BEEF CATTLE FARMS

Farm	Occupation of arable land	Occupation of pasture land (non arable)	Grain use	Human edible energy	Human edible protein
	m^2 / kg boned beef	m^2 / kg boned beef	kg / kg boned beef	MJ / kg boned beef	kg / kg boned beef
NE QLD Boned Beef	-1.9 ± 12%	1393.6 ± 10%	-0.29 ± 3%	-8.3 ± 4%	-0.13 ± 4%
SW QLD Boned Beef	4.9 ± 14%	2180.4 ± 16%	1.05 ± 6%	10.4 ± 6%	0.05 ± 6%

One useful metric of resource use and completion for human edible energy and protein comes from the ratio of human edible energy and protein consumed:produced. Both supply chains used some grain that could be diverted to the human food supply chain. However, this was offset (partially or totally) by production of co-products that replace human edible soy and canola. In the case of the NE supply chain, this resulted in ‘negative’ use of human edible inputs. Results for the NE supply chain could not be presented as a meaningful ratio, because human edible energy and protein consumption was negated by co-products. However, if the effect of co-products was removed, the ratio would be strongly positive for energy and protein. Beef production in the SW supply chain had a ratio of 1:1 for energy (one unit consumed per unit produced) and a ratio of 1:4 for protein (one unit consumed per five units produced). Considering this supply chain utilised grain finishing, the positive ratio for protein still highlights the net contribution of red meat production to overall human edible protein supply. This is because throughout the lifetime of the animal, most time is spent grazing grass in rangeland areas, on non-arable land unsuitable for other food production systems.

6.1.2 Environmental Impacts

Post processing environmental impacts generally increased (compared with results presented for live weight) in line with the mass reduction of the products, with small offsets from co-product substitution processes. Contributions to GHG emissions from meat processing arose from fossil fuel energy use and effluent treatment (methane), though both sources were minor compared to impacts from the farms.

TABLE 34 – ENVIRONMENTAL IMPACTS FROM PRODUCTION OF BONED BEEF FROM TWO QLD BEEF CATTLE SUPPLY CHAINS

Supply Chain	Soil depletion Potential	Soil Carbon Flux Potential	GHG emissions
	kg / kg boned beef	kg CO_2-e / kg boned beef	kg CO_2-e / kg boned beef
NE QLD Boned Beef	83.7 ± 43%	0.03 ± 84%	28.2 ± 15%
SW QLD Boned Beef	10.6 ± 80%	0.26 ± 16%	24.5 ± 16%

6.2 Beef Consumed at Home

The results for this stage are presented per kg of beef consumed in the home (either in Australia or Japan). Results presented at this point in the supply chain extend beyond production and processing to shipping, retail, use at home and wastage.

6.2.1 Resource Use

Energy use was significantly higher for the NE compared to the SW supply chain. Impacts from shipping, supermarket, home and wastage were particularly evident for energy demand. These stages contributed 58% and 51% of the total energy demand for the SW and NE supply chains respectively. The contributions varied between the supply chains, with shipping (Australia to Japan) contributing 18% to energy demand for the NE supply chain. Retail contributed 13-19% of total energy consumption, while energy consumption at the home ranged between 16 and 32% of total energy demand. The difference in energy demand in the home between Japan and Australia was partly because of the longer cooking times for meat in Australia. Figure 13 shows the energy demand for beef consumed in the home for the NE and SW supply chains.

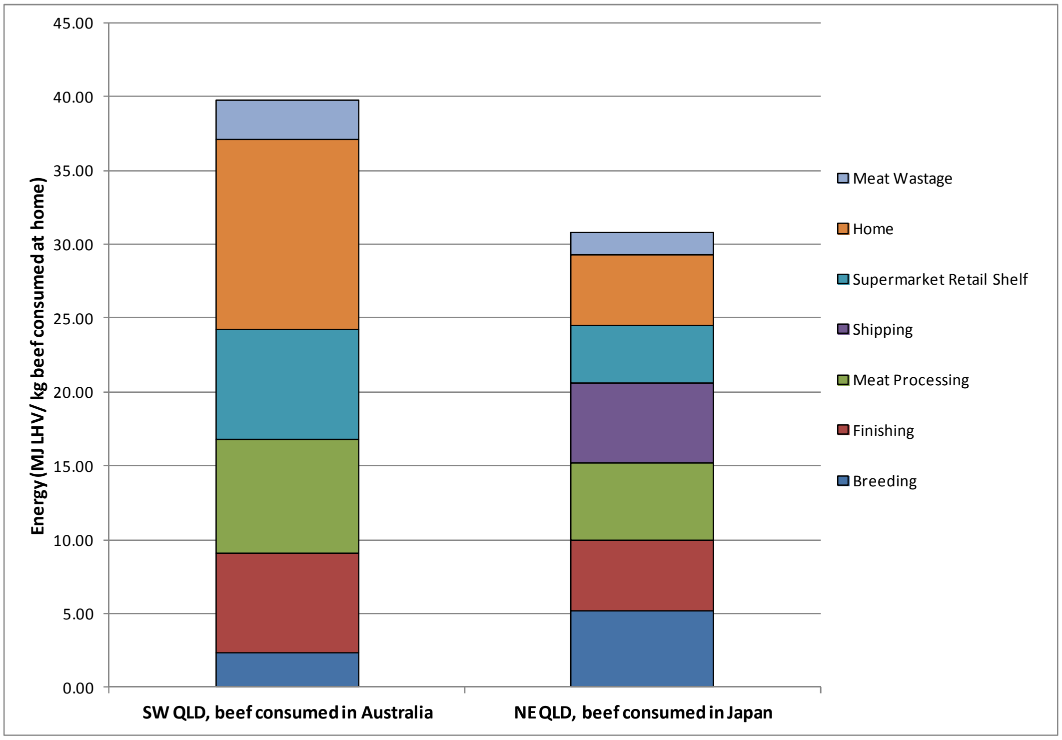

FIGURE 13 – ENERGY USE FOR BEEF CONSUMED IN THE HOME IN AUSTRALIA AND JAPAN

Consumptive water use for the post-processing supply chain contributed 7-10% of total water use, with the majority of this being related to meat wastage (resulting in overall increases in impacts across the supply chain). Figure 14 shows the consumptive water use for beef consumed in the home for the NE and SW supply chains. Total water consumption from meat processing was negative – this was due to the effects of using system expansion for the co-products which results in water being saved from displacing irrigated soybean.

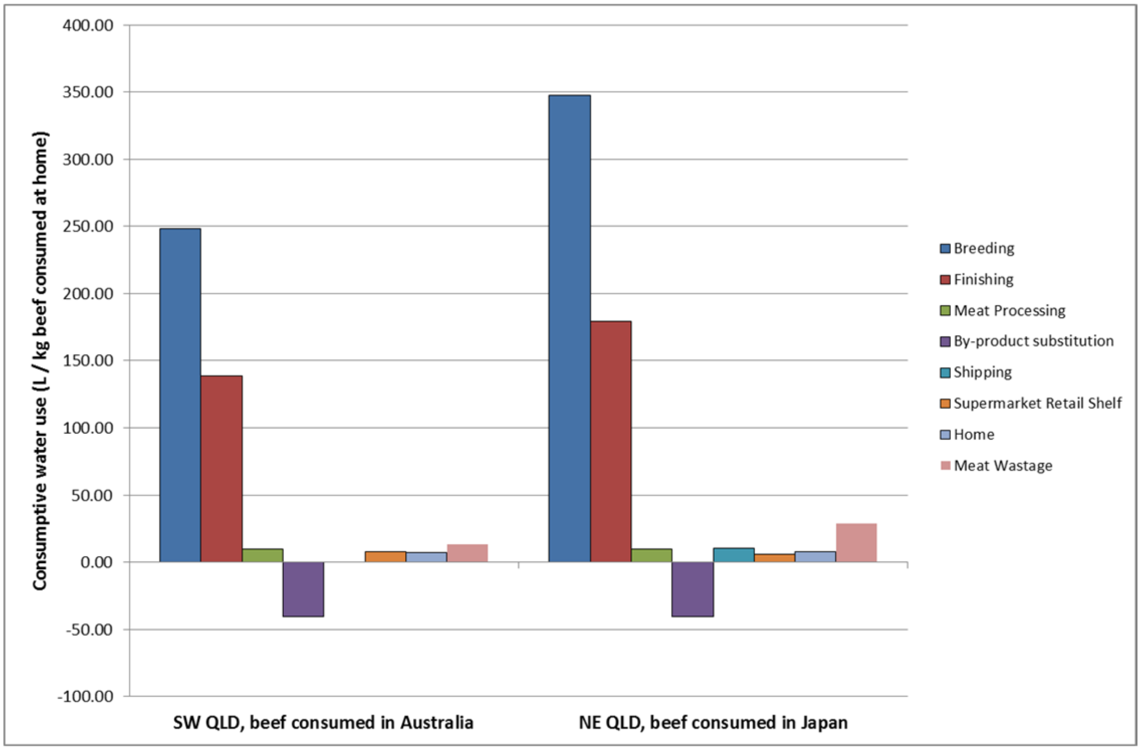

FIGURE 14 – CONSUMPTIVE WATER USE FOR BEEF CONSUMED IN THE HOME IN AUSTRALIA AND JAPAN

6.2.2 Environmental Impacts

Greenhouse gas emissions were 31.8 kg CO_2-e / kg beef consumed for the NE supply chain compared to 30.8 kg CO_2-e / kg beef consumed for the SW supply chain. Impacts from shipping, supermarket, home and wastage contributed 21% and 11% of the total GHG emissions for the SW and NE supply chains respectively. Meat wastage in the home was assumed to be 5% higher in Australia than Japan, which was the main difference between the two supply chains compared to boned beef results.

The same trend can be seen with the GHG emissions at the home – these ranged between 0.9% and 4.2% for the two supply chains, with the variation a response to higher meat wastage estimates for Australia compared to Japan. Retail of beef contributed 1.1 -2.5% of total emissions. Shipping contributed only 1.3% of the total GHG emissions for the NE supply chain, despite the relatively higher contribution to energy demand. Figure 15 shows the GHG emissions for beef consumed in the home for the NE and SW supply chains.

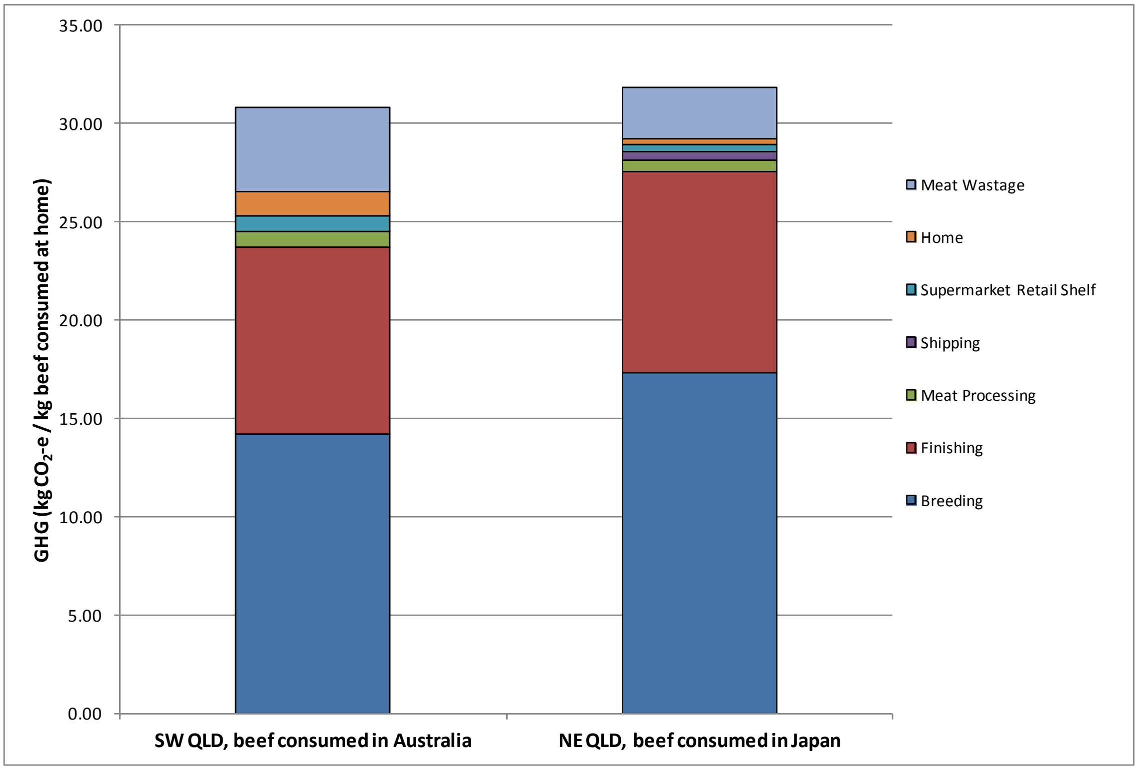

FIGURE 15 – GREENHOUSE GAS EMISSIONS FOR BEEF CONSUMED IN THE HOME IN AUSTRALIA AND JAPAN

7 Discussion

This study represents the first comprehensive LCA of Queensland beef supply chains from paddock to plate, covering a wide range of impacts relevant to the industry. A number of beef LCA studies have previously been completed in Australia and world-wide from which to draw comparisons with Australian production. The majority of these studied GHG emissions only, though some were found that investigated a wider range of impacts. Direct comparisons between LCA studies are difficult because of differences in system boundaries, handling of co-products, GHG and water inventory methods and impact categories. However, results from other studies are useful for indicative purposes provided differences are taken into account. This discussion was based on farm gate (live weight) results, because most studies covered the supply chain to the farm-gate only. Some studies reported their results on a carcase weight basis and where necessary, we converted results to a live weight basis using data from the studies to aid comparison.

7.1 Resource Use

The results from our study are generally lower in energy use and similar in GHG emissions intensity than other studies in the literature. Energy demand in our study ranged from 4.3-4.7 MJ / kg LW. In comparison, energy demand was 22.4 MJ / kg LW for ‘purpose grown’ beef production in the UK (Williams et al. 2006), 38-48 MJ / kg LW for feedlot or grass finished cattle in the USA (Pelletier et al. 2010) and 36-40 MJ / kg LW for beef production from France (Nguyen et al. 2012). Energy demand was lower where a proportion of beef was sourced from dairy herds, as shown by the national average value (15 MJ/kg LW) for UK beef reported by Williams et al. (2006). The lower energy use in our study reflects the low inputs for Australian beef production systems compared to Europe.

7.1.1 Water Use

Consumptive water use results in the present study were similar, though slightly higher than those reported by Wiedemann et al. (2012) and Ridoutt et al. (2012). All three studies applied the same temperature dependent drinking water prediction model, which resulted in higher predicted water intake for the Queensland farms in the present study compared to NSW or Victorian farms. Eady et al. (2011) also predicted water use for cattle produced on two Queensland properties, though it was not clear what methods were applied to generate the results. Excluding the influence of irrigation in some of the farms investigated by Wiedemann et al. (2012) and Ridoutt et al. (2012), the higher water use in our present study is mainly the result of higher predicted evaporation rates from farm dams in the present study. Ridoutt et al. (2012) estimated water use for beef production in six NSW regions, based on averaged data from government extension agencies. The volume of water stored in dams to supply annual drinking water demand was based on an assumed ratio of demand to supply. This volume of stored water to annual water demand (ratio) was found to be quite important to the prediction of total evaporation in our study. Total water stored influenced dam surface area and had a strong bearing on the ratio of storage evaporation to drinking water extracted. Drinking water use is an essential livestock input and cannot be readily modified. However, the efficiency of water supply and provision on a farm can be improved. Where water does not need to be stored in a dam or open surfaced tank, losses are negligible. However, losses can be considerable from open storages where evaporation rates are high, as is the case in Queensland. These losses may be mitigated by increasing dam depth, thereby reducing the surface area to storage ratio.

Ridoutt et al. (2012) also estimated stress weighted water use for six NSW regions. Stress weighted water use ranged from 3.3-221 L H_2O-e / kg LW, which was a broader range than found in our study. However, Ridoutt et al. (2012) included regions where water stress was higher than the supply chains we studied, and also included some farms where irrigation was used. Results for each of these studies are shown in Table 35.

7.1.2 Land Occupation

Land occupation could only be compared as 'totals' which are of limited value because no studies in the literature report arable and non-arable land separately. As expected, land occupation was much higher in the present study (656-1037 m^2) than most studies in the literature, in response to the low stocking densities on the farms studied. Wiedemann et al. (2012) found total land use to range from 87-100 m^2 / kg LW for a range of farming systems in NSW. Land occupation was shown to be considerably lower in Europe (~22-26 m^2 / kg LW, Nguyen et al. 2012; Williams et al. 2006) and the USA (84-120m^2 / kg LW, Pelletier et al. 2010). Assessment of total land occupation is not informative however, because it offers little insight into the resource value of this land, particularly when compared to other potential uses. We preferred to differentiate between arable and non-arable land resources. This is particularly relevant for discussions that relate to alternative food production systems (beef compared to poultry or vegetable proteins) because the land required for monogastric livestock (pigs and poultry) or vegetable protein production is not interchangeable with most grazing land in Australia.

7.2 Environmental Impacts

7.2.1 Eutrophication Potential

Assessing eutrophication potential for beef cattle production in the northern regions of Australia is constrained by the available research, and the lack of regionalised characterisation factors suitable for the production systems and aquatic ecosystems present. The NE property is located in a coastal catchment where eutrophication is a concern. However, direct nutrient losses were not likely from the farm because of the very low nutrient inputs and high levels of ground cover maintained. However, soil losses may also contribute to nutrient loads, and this requires further consideration for the industry in this region. The SW property is located in the far-northern reaches of the Murray Darling Basin. Eutrophication is a concern in this river catchment; but extensive research suggests that the predominant nutrient source is from gully and stream bank erosion rather than from fertiliser or surface runoff sources (Davis & Koop 2006). Considering the nutrient loss risk was very low for this farm also, the eutrophication potential for beef cattle production in this region is expected to be very low. This may be a contrast to parts of southern Australia, where nutrient runoff levels may be higher from grazing land (see Drewry et al. 2006), and where these nutrients may be more concerning for freshwater ecosystems.

7.2.2 Land Use Impacts

No studies were found in the literature that investigated the impacts of beef production on land occupation specifically, though Peters et al. (2011) did report nutrient flows and soil acidification at the farm level. Soil Depletion Potential (soil erosion) was shown to vary considerably between the two farms, ranging from 5-39.4 kg / kg LW for the SW and NE farms, mainly in response to differences in soil type and the corresponding susceptibility to erosion.

Soil Carbon Flux Potential ranged from 0.01 to 0.12 kg CO_2-e / kg LW for the NE and SW farms respectively. These positive values indicate potential losses of soil organic matter, and small contributions to GHG emissions. This assessment was subject to a high degree of uncertainty and carbon flux potential from pastures is debated among scientists. We assumed that soil carbon flux under native pastures was static, and that carbon losses (positive values) arose from cultivated land use, after Dalal & Chan (2001). The positive flows from the farms were associated with grain use.

7.2.3 Greenhouse Gas Emissions

Greenhouse gas emissions for the NE and SW QLD farms ranged from 12.9-11.2 kg CO_2-e / kg LW. European studies that investigated 'purpose grown' (i.e. non-dairy) beef production reported impacts in the order of 11-16 kg CO_2-e / kg LW. Casey & Holden (2006) reported GHG intensity of 11.1-13 kg CO_2-e / kg LW for Irish beef production, while Williams et al. (2006) reported 14 kg CO_2-e / kg LW for UK purpose grown beef. Similarly, Edwards-Jones et al. (2009) reported 16.2 kg CO_2-e / kg LW for beef production in Wales, though these authors also reported an extremely high value (48.6 kg CO_2-e / kg LW) for one case study farm where very high nitrous oxide emissions arose from soils. European studies that included beef from dairy herds reported lower GHG emissions. Cederberg et al. (2009b) reported 10.9 kg CO_2-e / kg LW as a national average for Swedish beef, while Williams et al. (2006) reported 8.7 kg CO_2-e / kg LW for 'average' UK beef, including beef from dairy enterprises. Outside Europe, Beauchemin et al. (2010) reported 13.8 kg CO_2-e / kg LW for a Canadian, feedlot finished production system, while Pelletier et al. (2010) reported 14.8-16.2 kg CO_2-e / kg LW for feedlot finished beef in the USA, and 19.2 kg CO_2-e / kg LW for grass/forage finished beef in the USA. Cederberg et al. (2009a) reported a national average emission for Brazilian beef of 15.4 kg CO_2-e / kg LW.

The GHG results of the present study are in general agreement with previous Australian studies. Only one other LCA study has been completed for Queensland production (Eady et al. 2011) and emissions from this study were slightly higher, largely because Eady et al. applied the methane prediction equation from Kurihara (updated by Hunter 2007) which estimates significantly higher emissions than the updated method from Kennedy & Charmley (2012). Consequently, the differences are more a response to differences in prediction methods than actual differences between the farms.

Three previous studies have been completed for southern beef production. These studies tended to investigate farms with slightly higher productivity (resulting in lower GHG) than the NE supply chain. Ridoutt et al. (2012; 2011) based their study on production and input data from NSW DPI gross margins (not case study farms) with fairly high levels of herd productivity (weaning rates and growth rates to slaughter), resulting in lower emissions than found from our NE study.

The results of the scenarios presented in this report for Queensland beef production are in agreement with Wiedemann et al. (2012), who found that grain finishing could reduce GHG emissions compared to grass finishing because of the higher growth rates in slaughter cattle.

TABLE 35 – GREENHOUSE GAS AND WATER USE FOR BEEF PRODUCTION – A COMPARISON WITH OTHER AUSTRALIAN STUDIES

Region	Class of cattle	GHG (kg CO_2-e / kg LW)	Consumptive Water Use (L / kg LW)	Reference
NE Queensland	Export, Japan Ox	12.9	248	This study
SW Queensland	Domestic, grain finished	11.2	183	This study
Nth NSW, Sth NSW, SE VIC	Domestic Grass/forage finishing	11.7-12.4	121-161	Wiedemann et al. (2012b)
Averaged NSW backgrounding	Grain finishing (Domestic market, 63d)	12.1	140	
Nth NSW, Sth NSW, SE VIC	Mid weight grass/forage finishing	11.3-13.0	107-298	
Nth NSW, Sth NSW, SE VIC	Mid weight grass/forage finishing – Drought conditions	14.3-14.8	144-262	
Averaged NSW backgrounding	Grain finishing (Mid-fed, 115d)	10.3	112	
Nth NSW, SE VIC	Heavy grass/forage finished bullocks (700 kg LW)	11-7-12.7	91-123	
Sth NSW	Heavy grass/forage finished bullock (600 kg LW)	13.4	131	
Averaged NSW backgrounding	Grain finishing (Long-fed, 335d)	11.8	111	
Sth NSW, SE VIC, inc. meat processing	Various grass fed	8-9.6 [a] (6.1 excl. breeding)	32 [a] (22 excl. breeding)	Peters et al. (2010a, b)
Sth NSW, inc. meat processing	Grain finishing (Mid-fed, 115d)	8.0-8.2 [a]	375-435 [a]	
Central NSW	Yearling (domestic grass)	10.4-10.6	24.7 - 167	CSIRO (Ridoutt et al. 2011; 2012)
Hunter and Central Western NSW	Mid weight and heavy grass finished steers	10.2-10.8	53.5-234	
Walgett-Gunnedah-Quirindi Casino-Glen Innes, Rangers Valley	Grain finished	10.1 (Quirindi) 12.7 (Rangers Valley)	160 139	
Gympie, QLD	Weaners (only)	17.5 – 22.9	118-155	CSIRO Eady et al. (2011)
Arcadia valley, QLD	Jap Ox - grass-fed	11.6 – 15.5	51.1-87	

[a] Results have been converted to a LW basis using dressing percentage and allocation results from the original study. Meat processing data could not easily be removed because insufficient data were supplied.

7.3 GHG Mitigation Potential

We investigated a range of mitigation scenarios that could be applied on the case study farms or more broadly in the Queensland cattle industry. In addition to GHG mitigation potential, we investigated trade-offs with other impacts such as energy demand, grain use, arable land use and water use. The mitigation strategies focused on three approaches: i) improving herd productivity via higher weaning rates (NE farm) and growth rate in slaughter cattle, ii) changing carbon sequestration rates by intensifying production and allowing additional land to regenerate, and iii) utilising pastures (legume) and supplements (oil) to reduce enteric methane emission rates.

The mitigation potential varied from quite modest improvements (<10%) for the SW supply chain, to >20% for the NE supply chain. Improvements were contingent on successfully establishing Leucaena pastures and achieving quite high levels of productivity on both farms, which may be difficult in practice. Consequently they may represent a maximum improvement potential rather than a practical improvement potential. The main difference between the supply chains was because there was a greater mitigation potential at the NE farm, which had lower initial weaning rates and a long grow out period for slaughter cattle on pasture.

For the industry scenarios, a number of options were shown to reduce GHG emissions compared to a standard, low input grass finishing operation targeting the Japan Ox market. Similarly to Wiedemann et al. (2012), grain finishing reduced GHG emissions, though the mitigation potential was greater in the present study (22-31%) largely because growth rates for our reference grass finished scenario was lower than most of the NSW farms studied by Wiedemann et al. (2012). We also modelled scenarios that involved improved growth rates in the backgrounding phase in the present study, while Wiedemann et al. (2012) looked only at the finishing phase. We found that there is potential to mitigate GHG by using Leucaena forages, resulting in lower per head enteric emissions and higher growth rates than grass only pastures. We also investigated the potential for carbon sequestration in both soils and the Leucaena trees and the influence on net GHG, which was found to result in substantial potential offsets.

To achieve the modelled GHG mitigations, it generally required other resources to improve productivity. Grain feeding utilised higher amounts of fossil fuel energy, higher areas of arable land and grain. Leucaena finishing was modelled using an irrigated system (required for year round backgrounding and finishing at high growth rates) which utilised high amounts of water and arable land. Dryland Leucaena was the only system that did not require significant additional resources compared to pasture finishing, and this scenario had a reasonable mitigation potential when sequestration sources were accounted for. It should be noted that sequestration potential in soil and vegetation may be difficult to quantify in commercial situations and may vary. The modelling presented here was based on measurements of Australian Leucaena pastures by Radrizzini et al. (2010), but the applicability of these results across Queensland is not known. None-the-less, the results broadly indicate that the sequestration potential from these sources may be large enough to be considered in greater detail.

7.4 Co-Product Assumptions

Modelling assumptions used to determine the impacts attributable to co-products and primary products may influence the final results significantly. In this study, handling of co-products at the point of slaughter was the most sensitive aspect, and to investigate the sensitivity of decisions made two alternative methods of handling co-products were analysed and are reported here; allocation on the economic value of products and co-products, and allocation on the mass of products and co-products.

In this study we applied a novel hybrid mass-system expansion approach, where meat, hides and human edible offal was handled using mass allocation (equal burden to each product) and system expansion was used to handle minor by-products such as meat meal, tallow and pet food.

We found that, compared to the hybrid system expansion approach, economic allocation resulted in 7%, 8% and 15% higher burdens applied to the meat product for GHG, energy and water respectively. The variance across the impact categories corresponds to the differences in the method applied; system expansion used soybean meal as a substitute product, resulting in a proportionally greater offset of water because of irrigation use in soybean production. In contrast, a mass allocation process (where the burden was allocated equally over the mass of all final products) resulted in 10-26% lower impacts, with the smallest differences being with energy and the largest being with GHG.

Further investigation into methods for handling co-production at the point of slaughter are warranted. In particular, an investigation of the proportion of protein partitioned to each product would be worthwhile.

8 Conclusions

This study is the first comprehensive LCA of Queensland beef production throughout the whole supply chain to the point of consumption. The use of case study data means that the results could not be considered representative of Queensland beef production in general, as they reflect the natural resource base and management of specific farms studied. However, for impacts such as water use and energy use (where 'industry average' data are difficult to obtain) this was considered the more reliable approach than desk-top modelling alone. One limitation in the study was that we did not include any herds with low growth rates. In some northern and western Queensland regions growth rates are known to be considerably lower than the farms studied here. With these limitations stated, a number of useful and important findings have come from this report.

The results show that Queensland beef production was relatively efficient with regard to resource use (low energy demand, low arable land use and slightly higher water use) and generated environmental impacts at similar or slightly higher levels than equivalent beef production from southern Australia. Compared to specialist beef production in Europe or examples from North America, energy demand tended to be lower and GHG emissions were similar. The similar GHG emissions were unexpected. In general, enteric methane emission intensity is quite high in Australia because herd productivity is not high. However, emissions from other sources (nitrous oxide, carbon dioxide) are very low. Nitrous oxide emissions are lower across most Australian agricultural industries because of the lower and less frequent rainfall, and lower levels of nitrogen cycling in Australian agricultural systems compared to Europe. Carbon dioxide emissions are low in response to the relatively lower energy demand. This is the first LCA study of Queensland beef to utilise methods for estimating enteric methane that are based on comprehensive Australian research. The result of using this updated method was a ~20% reduction in GHG compared to the previous Australian specific method.

This study applied a comprehensive method for assessing consumptive water use throughout the supply chain. This study and two studies of southern Australian beef production each report consumptive water use of <300 L / kg LW. In contrast, water footprint/virtual water results commonly quoted for beef are in the order of 15,000-100,000 L / kg beef. The large difference relates to the way in which rainfall is handled in the study. We chose to follow commonly accepted approaches to understanding water use by considering only flows from stored fresh water, as is found in a river, dam or bore for example. We excluded water that is evaporated or transpired through plants where this water was directly sourced from rainfall (so called 'green water'). Green water, or soil stored moisture for pasture or crop production, has very different characteristics in terms of transferability: it can't be used for anything other than plant production. Green water is also causally linked to land characteristics, and cannot be transferred even between many plant species because of soil or landscape constraints. For this reason, our study investigated impacts associated with green water as part of land occupation rather than as part of water resources. Importantly, the impact of using green water is minimal. Consumptive water use, in contrast, is a measure that reflects with the way water resources are legislated, traded and commonly understood in Australia.

Consumptive water use could restrict the water from being used for alternative purposes or the environment. However, it is not clear that simply 'using' a given volume of water generates negative environmental impacts, as this is more constrained by the level of water stress within a given catchment. In order to further investigate the impact of water use, the stress weighted water use indicator (a measure of the intensity of water use against a global water stress index) was applied. For both supply chains, stress weighted water use was lower than consumptive use, indicating the impacts of using water in these regions is not high. Considering the prevalence of water footprint data and the ongoing notion that beef cattle consume large quantities of water in the production system, this issue will require concerted, on-going efforts to re-educate the media and general public.

The assessment of land occupation focused on arable land resources, which are the most limited land resources in Australia and globally. We found that Queensland cattle utilised very small areas of arable land. Rangeland land occupation was high in comparison with other studies. However, land occupation by grazing minimally disturbed, native pastures in rangeland areas cannot be taken as a proxy for negative impacts on biodiversity. This land resource is also unsuitable for alternative food production purposes, such as grain (either for direct human consumption or for feeding to monogastric livestock).

We investigated the consumption of human edible protein and energy to produce beef, as a measure of the net food production from the beef sector. Ruminant livestock fed entirely on grain will have a relatively poor conversion efficiency for feed inputs to outputs. However, the cattle investigated in this study (even those finished with grain) consume very little of their total feed requirements as grain. The result is that the net production of human edible protein was considerably higher than the amount consumed throughout the system, demonstrating a net contribution to food production. Interestingly, because co-products from meat processing (such as tallow and meat meal) displace human edible energy and protein products, the grass finished cattle were found to generate more human edible protein not only via the primary meat product, but also via the displacement of soy and canola by co-products.

The post-processing supply chain results identified that impacts were generally highest from the production and processing phases (water, land use, GHG) and that the impact of transport to Japan had little impact on GHG emissions. Energy use was quite high during the post-processing supply chain however, and this could be a focus area for retailers and consumers. Meat wastage at the retail and consumer levels resulted in higher impacts for beef throughout the supply chain, and reducing this would be a useful mitigation strategy for consumers.

9 Recommendations

This study highlighted the significance of a number of recent methodological advances in the area of water and GHG research. Additionally, the study identified a range of potential options for GHG mitigation and also investigated important trade-offs between GHG and resource use. New approaches such as division of arable and non arable land resources and inclusion of intermediate resources such as grain were also presented.

The results of this study provide useful information for dissemination throughout the research community, particularly with regard to GHG mitigation research. The use of updated methods for assessing enteric methane show that the impact from this source is slightly lower than previously thought. Enteric methane is still clearly the largest emission source from extensive cattle production however, and rightly remains the main research focus. We also applied an updated nitrous oxide emission factor for grain finishing in feedlots, which resulted in lower predicted emissions from this stage of the supply chain. This factor requires further validation research, though to date most research indicates emissions will be lower than the factor applied by the DCCEE. This improves the benefit of grain feeding as a GHG mitigation strategy. Few studies have considered the full range of GHG emission sources when assessing mitigations (though this is becoming more common) and fewer still have considered impacts on other resources when attempting to mitigate GHG. Some mitigation strategies need to be carefully assessed across a wider range of impact areas in order to assess whether overall environmental outcomes are improved, or whether the burden is simply shifted from GHG to another impact area. A similar broad assessment of impacts would be valuable for a larger range of potential GHG mitigation strategies being discussed as options for the carbon farming initiative.

This study and others completed recently show that consumptive water use for beef cattle production is considerably lower than the so called ‘water footprint’ of beef cattle. Unfortunately, water footprint values are still commonly quoted, and the industry must take a pro-active and long term approach to re-educating the media and general public over this issue. Indicators of the impact of water use (i.e. stress weighted water use) show that water use by beef cattle across a number of farms and regions in Australia is low. One area not covered in detail in the current study is the impact of land transformation on hydrology. In some ‘single issue’ impact areas such as carbon footprinting it has been recommended that land transformation from clearing vegetation be included in the GHG assessment for a product. If this was done in LCA it would be necessary to investigate also the impact on water use, because clearing of vegetation results in large increases in runoff, boosting water supplies downstream.

Considering the potential positive impacts of soil carbon sequestration, further investigation may be warranted to reduce the uncertainty in this estimate. Likewise, the contribution beef cattle farmers make to protecting vegetation and providing wildlife habitat could be incorporated into such a study as a means of highlighting the so-called ‘ecosystem services’ provided by the industry in a quantitative way.

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Appendix 1 – Farm and Feedlot Inventory Data

Uncertainty

All inventory data are reported with an indication of uncertainty. Uncertainty was determined using two methods; firstly, the pedigree matrix system (Weidema & Wesnaes 1996), was used for most inputs from the technosphere (i.e. electricity, fuel) and water inputs. The second approach used minimum and maximum values determined from the survey data, which were input using a triangular distribution in the modelling program SimaPro 7.3. This approach was taken for some flows between sub-systems (i.e. feed use) and for some important emission factors in the manure management system. These data are reported as a range (percentage +/- mean).

Farm Inventory Data

Farms use a range of inputs including energy for transport and farm operations, inputs for crop and pasture production (fertilisers, chemicals), and inputs associated with livestock (veterinary products, feed). Additionally, farms relied on a number of services such as accounting, banking and communications. All inventory data were collected over a 24 month period, with some production data collected over a three year period to reduce seasonal variations.

Transport data were collected for all transfers of materials and livestock within the supply chain. Major transport stages included livestock transfers and grain transport to the feedlot. Transport data were calculated as tonne kilometres and were classified according to truck type, using AustLCI transport unit processes. Staff transport to / from work was calculated from staff records and reported travel distances.

Farm inventory data are reported using an arbitrary value of 500 kg of LW produced by the property, and includes all impacts arising from production of the live weight. The 500 kilogram value was selected because this reasonably approximates the mass of a sale animal (cull cow or steer) from each farm. Table 36 and Table 37 show the inventory data for each for farms.

TABLE 36 – MATERIAL INPUTS AND OUTPUTS FOR NE QLD FARM

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	12.2	1.06
Dry lick (feed supplement)		kg	9.7	1.06
Protein (feed supplement)		kg	154.9	1.06
Other supplements		kg	5.2	1.06
Drinking water	Data collected from farm	L	71 897.4	1.48
Land Occupation	Data collected from farm			
Modified Grazing, Pasture (non-arable)		ha	4.7	1.20
Un-modified Grazing, Pasture (non-arable)		ha	30.4	1.20
Energy	Data collected from farm			
Electricity		kWh	35.3	1.01
Oil		L	0.4	1.01
Gas		kg	0.8	1.01
Diesel		L	44.5	1.01
Petrol		L	8.1	1.01
Transport	Estimated transport distances for cattle and farm commodities	t.km	59.3	
Herbicides	Data collected from farm			
Combined chemical use		g	29.4	1.06
Other inputs and services				
Veterinary services		$	9.2	1.92
Repairs/maintenance		$	49.1	1.92
Communication services		$	7.5	1.92
Insurance		$	5.2	1.92
Automotive repairs		$	8.9	1.92
Automotive registration		$	1.9	1.92
Accounting		$	2.6	1.92
Banking		$	1.3	1.92
MLA levy		$	4.5	1.92
Outputs				
Cull cows		kg	219.4	
Excreted Manure				
Manure N	DCCEE (2010)	kg	189.1	
Emissions				
Enteric methane	DCCEE (2010)	kg	234.1	

TABLE 37 – MATERIAL INPUTS AND OUTPUTS FOR SW QLD FARM

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	11.0	1.06
Dry lick (feed supplement)		kg	24.1	1.06
Hay		kg	36.8	1.06
Drinking water	Data collected from farm	L	56 105.5	1.48
Land Occupation	Data collected from farm			
Modified Grazing, Pasture (non-arable)		ha	3.3	1.20
Un-modified Grazing, Pasture (non-arable)		ha	56.1	1.20
Energy	Data collected from farm			
Electricity		kWh	12.6	1.01
Oil		L	0.6	1.01
Gas		kg	2.6	1.01
Diesel		L	25.0	1.01
Petrol		L	1.0	1.01
Transport	Estimated transport distances for cattle and farm commodities	t.km	28.9	
Other inputs and services				
Veterinary services		$	8.3	1.01
Repairs/maintenance		$	28.3	1.01
Communication services		$	4.3	1.01
Insurance		$	0.7	1.01
Automotive repairs		$	5.1	1.01
Automotive registration		$	1.9	1.01
Accounting		$	5.2	1.92
Banking		$	2.6	1.92
MLA levy		$	5.4	1.01
Contractors		$	8.5	1.01
Contract helicopter mustering		$	23.8	1.01
Licences and permits		$	3.1	1.01
Freight and cartage		$	0.5	1.01
Outputs				
Cull cows		kg	138.5	
Excreted Manure				
Manure N	DCCEE (2010)	kg	179.2	
Emissions				
Enteric methane	DCCEE (2010)	kg	216.4	

Feedlot Inventory Data

Feedlot inventory data were collected over a one year period from detailed metering and monitoring of energy use, commodity use and livestock numbers and performance. Manure production was estimated from feed and cattle performance data using the BEEFBAL model, and additional input data were collected from the feedlot managers as required.

TABLE 38 – MATERIAL INPUTS AND OUTPUTS FOR FEEDLOT

Inputs	Data source description	Units	per finished animal (347 - 437 kg LW)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	347.0	
Feed ration [a]	Data collected from feedlot	kg	600.1	1.06
Land Occupation	Data collected from feedlot			
Feedlot land use		m^2	3.5	1.20
Effluent irrigation area land use		m^2	7.4	1.20
Energy	Data collected from feedlot			
Electricity		kWh	3.2	1.01
LPG		L	0.1	1.01
Diesel		L	0.9	1.01
Petrol		L	0.04	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	71.8	
Other Purchases and inputs (expenses)				
	Veterinary services	$	7.3	1.48
	Communication services	$	0.1	1.48
	Insurance	$	0.1	1.48
	Automotive and feedlot infrastructure repairs	$	9.1	1.48
	Accounting	$	5.1	1.48
	MLA levy	$	5.2	1.48
	Horse feed	kg	0.3	1.48
	Staff travel	km	0.4	1.48
	Freight and cartage excl. livestock	tkm	0.01	1.48
Outputs				
Finished animal	Animal to abattoir	kg	437.3	
Excreted Manure				
Manure N	Mass Balance	kg	12.1	±10%
Manure VS	Mass Balance	kg	71.2	±10%
Manure P	Mass Balance	kg	1.4	±10%
Manure K	Mass Balance	kg	4.5	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979) and Beauchemin et al. (2008)	kg	9.5	

a – dry matter

Feed Milling and Rations

Feed milling inventory data were based on records kept by the feedlot. These data are presented in Table 39.

TABLE 39 – MAJOR INPUTS FOR FEED MILLING AT FEEDLOT

Inputs	Data source description	Units	Per tonne delivered to bunk
Energy			
Electricity	Data collected by feedlot	kWh	6.6
LPG		L	5.0
Diesel		L	0.8
Water			
Bore water	Data collected by feedlot	L	130.6
Transport	Est. transport distances for commodities to the feedlot	t.km	148

Feed inputs are the largest input for feedlot cattle production. Cattle are fed on diets matched to the nutritional requirements of the growing animals. Rations are formulated on a 'least cost' basis, resulting in variations to the input products throughout the year. For the purposes of the study, aggregated commodity inputs (aggregated over 12 months) were used. Feed input data were also required for modelling manure GHG emissions (i.e. digestibility, ash and crude protein) and these data were generated based on the specific rations. Commodity inputs to the rations were simplified using a substitution process (Wiedemann & McGahan 2011, Wiedemann et al. 2010b).

Data were not available for a number of minor dietary inputs. These inputs fall into two categories; products that require a low level of manufacturing and are of low cost (i.e. salt) and products that are high cost such as vitamins, synthetic amino acids and some minerals. High cost inputs are more likely to be associated with high levels of manufacturing (and energy input) and may be transported globally. To address this, low cost inputs were substituted for lime (calcium carbonate), and high cost inputs were substituted for synthetic amino acids using economic value to inform the substitution ratio.

Feed data were collected for the total feed intake over one year. Commodity inputs for the cattle rations were obtained from the feed mill and from the feedlot nutritionist. There are many rations fed throughout the year with a different formulation based on the nutritional requirements of the animals and the cost of inputs. To simplify these numerous rations, representative rations were developed for the feedlot. Table 40 shows the aggregated, simplified rations for the feedlot.

TABLE 40 – AGGREGATED, SIMPLIFIED RATIONS FOR THE FEEDLOT

Commodities (protein content in brackets)	Units	Amount
Barley (10%)	kg	40.4
Sorghum (10%)	kg	305.8
Wheat (13%)	kg	298.7
Cottonseed Meal	kg	7.1
White fluffy cottonseed	kg	91.8
Wheat Hay	kg	20.8
Wheat Straw	kg	17.4
Sorghum Silage	kg	132.0
Cotton Hulls	kg	14.9
Recycled Oil	kg	12.3
Feed additives	kg	58.8
Total	kg	1000.0

Background Data Sources

All processes that were part of the system boundary, but beyond the farm boundary, were included in the background system. These data were drawn from a number of inventory databases, in particular, the Australian AustLCI database and EcoInvent databases provided the majority of background process data. Upstream data associated with services such as repairs, telephone and veterinary services were based on financial records from the supply chain matched with economic input-output tables from the US economy. Impacts associated with services are typically very small, however this approach provided a comprehensive coverage of these impacts and was therefore included for completeness. No adjustment was made for conversion of Australian dollars to US dollars, as the services were not assumed to be driven by exchange rates.

Grain protein and energy levels were determined from typical analyses, with the energy content taken as the digestible energy rather than gross energy (i.e. the energy that could be harnessed by humans if used directly for food). Additionally, corrections were made for losses in the human edible grain supply system such as grain husks and brans which enter animal supply chains. These corrections were no more than 10% by volume for cereal grains (wheat) and were zero for some grains such as maize (corn).

Appendix 2 – Land Occupation and Nutrients

Land Occupation

Land occupation was divided into three classifications; i) arable land (land used for grain cropping, forage cropping or grazing during a pasture ley); ii) modified, non-arable grazing land (land that was cleared and in some cases sown with legume and grass species and fertilised with super phosphate (pasture improved), and iii) unmodified, non-arable grazing land (land that is utilised for grazing with minimal disturbance of the natural vegetation, with no added legume or pasture species, and no added fertiliser).

At each farm, the proportion of land in each category was determined from information provided by the farmers and from field observations. Land areas were accurately determined using GIS software and aerial photography or satellite imagery. For each land occupation type, pasture production and utilisation rates were determined through discussion with the farmer and from stocking rate records.

No characterisation factors were applied, and land occupation data were reported in m^2 of land occupied over a 12 month period.

Soil Depletion Potential

Erosion rates were determined using spatial data from the National Land & Water Resources Atlas (NLWRA 2001b). The main advantage in using the NLWRA data was the availability of a consistent dataset covering all the properties of interest, with estimates of pre-European (baseline) and post-European erosion rates. One disadvantage in this dataset was the coarse resolution of the mapping. Because of this, the NLWRA note that the data are not suitable for property scale assessment. To address this, additional data regarding ground cover and observed erosion was collected during site visits via field observations and through discussion with farmers. Additionally, a qualitative assessment of erosion was made for each farm based on visible signs of erosion from aerial photography or satellite imagery. An uncertainty factor (± 50%) was also applied to account for inherent uncertainty in the estimates. For both sites erosion rates were considered to be at the lower end of the scale identified by the NLWRA mapping. Data are reported in Table 41 to Table 44.

GHG Emissions from Runoff and Leaching

Nitrogen lost via leaching and runoff may contribute to greenhouse gas emissions (indirect nitrous oxide emissions). Two alternative approaches were used to estimate nitrogen losses from runoff; firstly, values were taken from the literature where similar sites were available. Provided data were reported on a mass basis (i.e. Ridley et al. 2003), these data were used directly. Where data were available only as a concentration of runoff, additional calculations were required to determine annual runoff. These estimates (necessary for the dam water modelling also) were based on annual rainfall and runoff fractions using the following equation:

$$\boldsymbol{Runoff\ (ML/ha/yr)} = \frac{\boldsymbol{Rainfall\ (mm)}}{\boldsymbol{1000\ (\frac{mm}{m^3})}} \times \frac{\boldsymbol{10\,000m^3}}{\boldsymbol{ha}} \boldsymbol{/yr} \times \frac{\boldsymbol{runoff\ frac}}{\boldsymbol{1000\frac{m^3}{ML}}} \qquad \text{EQUATION 1}$$

Runoff fractions were obtained from a series of reports by CSIRO. Where these data were not available, runoff was estimated using literature values for the region. The nutrient concentrations were then converted to values in kg/ha.yr using the following generalised formula:

$$\boldsymbol{Runoff\ N(kg/ha/yr)} = \frac{\boldsymbol{Runoff(ML/ha/yr) \times 10^{6}\left(\frac{L}{ML}\right) \times N_{conc}\left(\frac{mg}{L}\right)}}{\boldsymbol{10^{6}\left(\frac{mg}{kg}\right)}} \qquad \text{EQUATION 2}$$

It was assumed that no leaching occurred at either of the two sites and so the contribution of nitrous oxide emissions from leaching was zero. To determine the indirect nitrous oxide emissions which occur as a result of nitrogen loss in runoff, the predicted N runoff losses were multiplied by the nitrous oxide emission factor 0.0125 kg N_2O/kg N (DCCEE 2010).

Table 41 to Table 44 show the parameters used to calculate sediment loss and nitrogen loss in runoff for each of the farms investigated in this study.

TABLE 41 – NE QLD – MODIFIED GRAZING, PASTURE (NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	660	-	Hawdon et al. (2008)
Runoff fraction	-	0.09	-	Hawdon et al. (2008)
N in runoff and subsurface flow	mg/L	2.05	0.101-4.0	O' Reagain et al. (2005)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0.59	± 50%	NLWRA (2001a)

TABLE 42 – NE QLD – UN-MODIFIED GRAZING, PASTURE (NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	660	-	Hawdon et al. (2008)
Runoff fraction	-	0.09	-	Hawdon et al. (2008)
N in runoff and subsurface flow	mg/L	0.101	0.101-4.0	O' Reagain et al. (2005)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0.59	± 50%	NLWRA (2001a)

TABLE 43 – SW QLD – MODIFIED GRAZING, PASTURE (NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	551	-	CSIRO (2008)
Runoff fraction	-	0.05	-	CSIRO (2008)
N conc in river as a result of runoff and subsurface flow	mg/L	0.436	0.423-0.449	DERM (2012)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0.84	± 50%	NLWRA (2001a)

TABLE 44 – SW QLD – UN-MODIFIED GRAZING, PASTURE (NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference

Rainfall	mm	551	-	CSIRO (2008)
Runoff fraction	-	0.05	-	CSIRO (2008)
N conc in river as a result of runoff and subsurface flow	mg/L	0.423	0.423-0.449	DERM (2012)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0.84	± 50%	NLWRA (2001a)

Feedlot Data

At the feedlot, leaching was assumed to be negligible, because these facilities are designed and tested to strict standards in this respect (see Skerman 2000). Runoff is also controlled via construction of containment ponds to limit the loss of nutrient rich water to occasional (one in 10 year) overtopping events. Long term losses were averaged and taken into account in the indirect N_2O from runoff assessment.

At the feedlot, it was assumed that soil is lost from the land area which was irrigated with effluent from the pond. The parameters used to determine sediment loss and nitrogen loss in runoff for the feedlot are shown in Table 45.

TABLE 45 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS IN RUNOFF FOR FEEDLOT EFFLUENT IRRIGATION LAND OCCUPATION

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	533	-	BOM data
Annual Irrigation	mm	75	-	Averaged values
Runoff fraction	-	0.10	-	Averaged values
N in runoff and subsurface flow	mg/L	30	15-45	Averaged values
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	2.50	-	Prosser et al. (2002)

Cropping Processes

The nutrient losses which occurred as a result of the cropping process to produce the grains e.g. wheat, barley, sorghum etc., used in this study were based on a cropping process developed for the northern grain growing region (NSW north-east/west and QLD south-east/west regions). In order to determine the soil and runoff losses, and associated indirect N_2O emissions as a result of cropping, a review of the literature and available data was conducted in order to find studies which reflected the conditions in the northern cropping region. Rainfall data for six sites from 2000-2011 within this region was collected using the BOM website (BOM 2012). These six sites included three from Queensland (Roma, Taroom and Dalby) and three from New South Wales (Gunnedah, Narrabri and Wellington). The average rainfall for the six sites over the 11 year period was 595 mm with a range of 551-653 mm.

In order to determine the soil erosion rate for the cropping processes, it was first necessary to define the land management strategy. The National Land and Water Resources Audit (NLWRA 2001a) determined that average erosion rates for cereal cropping land (excluding rice) with no conservation practice was 2 t/ha/yr. However, in recent years there has been a shift from conventional tillage practices to zero-tillage within the northern cropping region. The Australian Bureau of Statistics (ABS 2009) reports that over 50% of the land in this region is now under zero-tillage. Erosion rates from zero tillage or low tillage cereal cropping is lower than conventional tillage, particularly where stubble is burned or removed. Littleboy et al. (1992) estimated erosion rates for sites at Gunnedah, NSW (1 t/ha/yr) and Dalby, QLD (3 t/ha/yr) using zero-tillage management practices for wheat production. The National Land and Water Resources Audit (NLWRA 2001a) suggest erosion rates under best management practices to be <1 t/ha/yr. For the present study, an erosion rate of 1 t/ha/yr was used.

The volume of runoff was used to determine nutrient losses from crop land. Runoff was averaged from estimates by Littleboy et al. (1992) for Dalby (59 mm) and Gunnedah (35 mm). Based on these values, the annual runoff as a fraction of rainfall was assumed to be 8%. This analysis allows for the determination of the indirect nitrous oxide emissions from N in runoff. Table 46 shows the parameter values and uncertainty ranges assumed for this study used to determine the sediment losses and nitrogen loss from runoff for the cropping processes.

TABLE 46 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FROM RUNOFF FROM LAND IN THE NORTHERN CROPPING REGION

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	595	551-653	BOM (2012)
Runoff fraction	-	0.08	-	Littleboy et al. (1992)
N in runoff and subsurface flow	mg/L	5.9	2.95-8.85	Murphy et al. (2011)
Soil erosion rate	t/ha/yr	1.00	0.5-1.5	Littleboy et al. (1992), NLWRA (2001a)

Appendix 3 – Water Use Inventory

Methodology

Inventory methods in LCA are closely linked to impact assessment. The key limitation to conducting a water balance or water footprint (both essentially inventory methods) is that neither give a clear indication of what impact will be caused by the water use activity. Inventory development in LCA has therefore focussed on refining the definitions of water use and determining what additional information is required to assess the impact of water use. Because global freshwater reserves are limited (at any given time) and subject to pressure, this is the focus of all investigations.

Water in LCA can be classified using the standard classification for abiotic resources, based on the regeneration potential. The three main types of freshwater resources thus classified include deposits, funds and flows (Koehler 2008).

Freshwater deposits represent:

- Non-replenishing groundwater stocks (which are finite resources) and are only very slightly replenished during the lifetime of a human
- Funds, which may be characterised as sub-artesian groundwater supplies, lakes or dams (exhaustible resources), which are naturally replenished as long as they are not irreversibly impaired
- Flows, which refer to streams and rivers (non-exhaustible in principle).

In addition to describing the source type, the term ‘use’ requires clarification. Owens (2002) provided a number of different classifications to differentiate between consumptive and non-consumptive uses, and between uses that result in depletion. These are:

- Water use – water is used off-stream and is subsequently released to the original river basin (downstream users are *not* deprived of any water volume).
- Water consumption of consumptive use – off-stream water use where water release or return does not occur (i.e. evaporation from a storage, transpiration from crop production).
- Water depletion – Withdrawal from a water source that is not replenished or recharged (i.e. a water deposit).

Bayart et al. (2010) provided a detailed framework for assessing water use in LCA at the inventory and impact assessment level. Their study proposed two categories of fresh water use:

1. Freshwater degradative use (water that is returned to the same catchment from which it was used, but with altered water quality)
2. Freshwater consumptive use (water that is not returned to the same catchment because it is evaporated, integrated into a product or discharged into a different catchment or the sea).

The authors consider both categories to be relevant for in-stream and off-stream uses. In-stream consumptive uses include evaporation losses from government managed water supplies, which will be relevant to an industry such as beef.

Bayart et al. (2010) also differentiate between “competition for fresh water use” and “freshwater depletion” in the following way. Competition for fresh water use refers to the situation where availability is temporarily reduced for current uses. Depletion refers to the situation where the amount of freshwater in a watershed and/or fossil groundwater is reduced. Depletion is said to occur when the rate of consumptive use exceeds the renewability rate over an extended period of time.

In order to differentiate water use using the above categories, Bayart et al. (2010) recommend that a water balance is used to populate the inventory. The balance should also distinguish resource type (i.e. groundwater, surface water) and water quality. Mila I Canals et al. (2009) likewise advocates determining consumptive water uses and water returns to ecosystems using a water balance.

Water quality is an important consideration in agricultural systems, particularly for discharge water. Bayart et al. (2010) did not investigate water quality in depth, but did note that two approaches could be used; i) quality could be assessed using a ‘distance-to-target’ approach, or ii) a functionality approach could be taken.

The distance-to-target approach would investigate the equivalent effort necessary to process a water output to the same quality as the water input. This could take into account additional water required to dilute nutrient levels to acceptable (i.e. river health) levels prior to release. Alternatively, it could take into account the energy required to purify a resource to the same quality. The ‘functionality’ approach is a means by which quality categories are established and water use is defined in terms of the water category for inputs and outputs.

These recommendations are comprehensive and logical, and provide a robust framework for developing water use inventories. However, there are no examples yet provided for Australian agricultural products that use these classifications.

An additional component of the inventory is the relationship between land occupation and water availability. When assessing the impact of an agricultural system, it is important to identify whether the system alters the flow of runoff to the environment as this is a component of water use. Milà i Canals et al. (2009) proposes a method whereby the difference in evapotranspiration between the system investigated and a reference system (i.e. natural vegetation) is used to determine the effect of the system on the water balance. Where a system evapo-transpires more water than the reference system, this results in additional water use that is attributable to the product grown on that land. Likewise, if a production system utilised less water than the reference system (as is often the case in Australia) a negative flow (or credit) may be applied.

Consumptive fresh water use represents the volume of fresh water used by a production system and is an inventory output from LCA. Inventories are best compiled using a water balance approach to define both inputs and consumption (outputs). Because of the widespread interest in water use, it is often reported as a result in LCA research. It is important however to extend this to investigate the impacts of water use on the environment using an impact assessment method.

Data Collection and Modelling Approach

The water inventory was developed by using a series of water balances for important processes in the foreground system. Full characterisation of water sources (inputs) and outputs from each stage were determined, including all losses. Depending on the method used, water use was based on either inputs (i.e. the ABS method) or outputs (the Consumptive Fresh Water Use method).

The main components for the foreground and background system are listed here.

Foreground system for farms:

- Livestock drinking water
- Drinking water supply system (Farm water balance 1)
- Irrigation water (where relevant)

Foreground system for feedlot:

- Feedlot pen (drinking) water
- Other feedlot water uses – feed milling, office and amenities, etc
- Feedlot water supply system (Feedlot water balance 1)
- Feedlot runoff capture (Feedlot water balance 2)

Background system for farms and feedlot:

- Water use in feed grain supply
- Water use in other inputs (i.e. energy)

Consumptive water use data for background processes are not well documented within the AustLCI and EcoInvent databases. Water use within background databases tends to be 'input water' only; consumptive and non-consumptive uses are not differentiated. These sources contributed only minor amounts of water to the system and in the absence of greater disaggregation in the databases, we assumed all input water to be consumptive. This may result in a small (<2%) over-estimate of water use.

Impact Assessment

The stress weighted water use impact assessment method applied different stress weighting factors for different regions of Australia where the farms and feedlot were located. To calculate the stress weighted water use, consumptive water use 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_2O-e; Ridoutt & Pfister 2010). Aggregated water use inventory data for the farms and feedlot are presented in Table 47.

TABLE 47 – SUMMARY OF SITE DATA USED IN WATER MODELLING FOR THE TWO FARMS AND FEEDLOT

System	Rainfall (mm / yr)	Pan Evaporation (mm / yr)	WSI
NE QLD Supply Chain	660	2154	0.110
SW QLD Supply Chain	551	2179	0.021
Feedlot	533	2012	0.021

For background products that may be sourced from many regions, we applied the same approach as Ridoutt et al. (2011a) by using the Australian average WSI value of 0.402 for these sources.

Farm Water Inventory

Modelling Livestock Drinking Water Use

Data were not available on the actual volume of water supplied for drinking on the grazing farms, and a measurement campaign was beyond the scope of this project. Estimation of water use at the farm level was complicated by the multiple sources used; i.e. bores, dams, creeks and reticulated supply, in varying proportions during the year.

Several factors determine drinking water intake for cattle, including feed intake, ambient and water temperature, class of animals and live weight (National Research Council 1996). Water use can be particularly variable in response to climate. The drinking water prediction equations from Ridoutt et al. (2011a) were applied in the present study.

Farm Water Supply Balance

Water supplies were from creeks, bores, reticulated supplies or on-farm storage dams. Table 48 shows the different sources for water supply at the farms, along with the proportion of total water supplied by each. These sources have different levels of supply efficiency. Supply efficiency relates to the losses incurred to supply a given quantity of water. For bore and reticulated supplies, losses on the case study farms were minimal. Water supplied from creeks was also considered to have minimal losses other than what would naturally be incurred in the absence of cattle farming. Farm dams that capture surface runoff however can have high loss rates associated with evaporation and seepage. Both of the case study farms used dams as a source of drinking water. The method used for determining water use from the supply system is influenced by the selection of the reference system and the ‘boundaries’ of the assessment. If the water supply system is considered for each farm as the difference between the presence or absence of farm dams, there will be losses from the supply system attributable to livestock production (i.e. evaporation from farm dams). However, if water use is assumed to be equivalent to the difference between the water balance for the current management system and the original reference land occupation (which in most cases would be open forest land) as per Mila I Canals et al. (2009), the water balance will in most cases be strongly positive, because pastures in Australia tend to have higher runoff rates than the original forest (see review by Brown et al.(2005)). In the present study we have considered the reference system to be pasture land in the absence of dams, rather than taking land use change from the natural state into account. This is a major distinction. However, land use change was outside the scope of the study and accounting for this would affect multiple impact categories.

An assessment of the water supply was made at each farm, based on records and input from the farmers and from an analysis of the property layout. Based on this analysis, the breakdown of water sources for the two farms was determined and is reported in Table 48.

TABLE 48 – SOURCES OF WATER SUPPLY FOR FARMS

Source of water supply	% of total water supply	
	NE QLD Farm	SW QLD Farm
Dam	42%	58%
Creek	33%	17%
Bore	25%	25%

Evaporation

Pan evaporation is the simplest way of estimating evaporation. The pan method involves taking a direct measurement of natural evaporation from a water surface, in a shallow pan. Evaporation pans are simple but they require daily measurement and maintenance and there may be significant variation between the evaporation from a small, steel pan and a large deep water body (Watts 2005). The calculation of open-water evaporation is achieved by applying a 'pan factor' to the measured evaporation. The equation for this conversion is:

$$\boldsymbol{E = K_P \times E_{Pan}} \qquad \text{EQUATION 3}$$

where:

E = open-water evaporation (in mm/day)

K_p = pan factor, constant determined by the pan siting, relative humidity and wind speed.

E_{pan} = pan evaporation (in mm/day)

The value of K_p can vary widely. Ham (1999) determined a value of 0.81 for a farm lagoon containing animal waste. Ham (2007) showed the ratio between lagoon and pan evaporation was variable but typically was between 0.7 and 0.8. In the present study, a K_p value of 0.8 has been applied for determining evaporation from water storages.

In addition to evaporation losses, dams may also lose water via seepage through the bank or floor of the dam. Seepage is considered a marginal contribution to total storage losses when compared to evaporative losses and has been the focus of limited research. Dam seepage may in some instances flow to groundwater or surface water. In other instances it may evaporate. In this project, losses via seepage were considered to be a non-consumptive transfer rather than a use and were therefore not attributed to the product. The efficiency (measured as a ratio of losses to water supplied) for dams on each farm and feedlot is reported in Table 49. Ratios differ based on evaporation rates, dam surface area to volume ratios, and as a function of the utilisation rate of the dams. Farms that had more dams to improve reliability of supply in very dry years necessarily lost higher volumes of evaporation because of the large volume of water stored annually but not utilised.

TABLE 49 – EVAPORATION AND SEEPAGE SUPPLY EFFICIENCY FACTORS

Inflows	NE QLD	SW QLD	Uncertainty (SD)
Dam storage to annual drinking water demand ratio	4.6	5.2	-
Net evaporation to drinking water supply ratio [a]	1.92	1.41	1.45
Seepage to drinking water supply ratio [b]	0.81	0.51	1.45

[a] L evaporated per L supplied. [b] L seepage loss per L supplied

Feedlot Water Use Activities

In the feedlot, water is primarily used for drinking and cleaning. It is very difficult to disaggregate these water 'uses' at a commercial feedlot. Hence, to establish a water balance a number of assumptions were required to quantify uses and outputs.

Feedlot Drinking Water Use

Drinking water at the feedlot was predicted based on the work by Winchester & Morris (1956). This study provides predictions of daily water intake for both *Bos taurus* and *Bos indicus* cattle. Winchester & Morris related water intake per day to ambient temperature, dry matter intake (DMI) and breed. Their trials were conducted in a constant temperature chamber.

Results showed that up to an ambient temperature of 30^{o}C, the rate of water consumption per unit dry matter intake remained fairly constant. As the temperature exceeded this level, consumption rose dramatically due to increased evaporative (cooling) demand. Winchester and Morris (1956) measured actual water intakes of 16 L/kg DMI per day by *Bos taurus* breeds, and about 10 L/kg DMI per day for *Bos indicus* breeds.

Watts et al. (1994) developed the following relationships from the collated data of Winchester and Morris:

$$\boldsymbol{Bos\,taurus} \quad \boldsymbol{WI = DMI \times \left(3.413 + 0.01592e^{0.17596T}\right)} \qquad \text{EQUATION 4}$$

$$\boldsymbol{Bos\,indicus} \quad \boldsymbol{WI = DMI \times \left(3.076 + 0.008461e^{0.17596T}\right)} \qquad \text{EQUATION 5}$$

Where:

WI = water intake (litres/head/day)
DMI = dry matter intake (kg DM/head/day)
T = ambient temperature (degrees Celsius)

Using these relationships, this study found that the drinking water consumption at the feedlot over a twelve month period was equivalent to 43.1 L/head/day. This prediction compares favourably with the results of the study by Davis et al. (2008a), which investigated the actual drinking water consumption at eight different feedlots in Australia. The average drinking water consumption across all feedlots for March 2007 to February 2008 ranged from 31 L/head/day to 46 L/head/day, with an average in the order of 40 L/head/day.

Water intake with feed and cattle

In addition to drinking water, cattle ingest a small amount of water with feed equivalent to the moisture content of the feed (generally around 10-20%) and generate additional water from the breakdown of carbohydrates, fat and protein in the feed (metabolic water).

Water ingested with feed was determined from the analysis of diets provided for the feedlot multiplied by the average feed intake of cattle. Metabolic water was determined using the simple relationship reported in the National Research Council (1996), which suggested 0.6 L of water is produced per kilogram of feed.

Water inputs also arise from cattle entering the feedlot, which contribute to the water balance from the proportion of water in the body mass of the animals. Water content in cattle was assumed to be 36% of body weight, based on Ridoutt et al. (2011a).

Feedlot Water Supply

Water was supplied at the feedlot from a bore (20% of supply) and from a series of farm dams supplying the remaining 80%. Evaporation and seepage were determined from the dams to calculate total consumptive water use (Table 50).

TABLE 50 – FEEDLOT DAM EVAPORATION AND SEEPAGE SUPPLY EFFICIENCY FACTORS

Inflows	SW Feedlot	Uncertainty (SD)
Net evaporation to drinking water supply ratio [a]	1.41	1.45
Seepage to drinking water supply ratio [b]	0.51	1.45

[a] L evaporated per L supplied. [b] L seepage loss per L supplied

Water Loss Pathways from Cattle

Water losses or outputs from the cattle herd are in the form of water uptake in live weight gain, losses via respiration and perspiration, and excreted losses via urine and faeces.

Water contained in the live weight of sale cattle was determined using a 36% moisture content as used by Ridoutt et al. (2011a). Evaporative water loss from cattle (respiration and perspiration) is a function of DMI and mean temperature and was determined using the equation reported by Ridoutt et al. (2011a):

$$\boldsymbol{W_{evap\,loss} = C + (W_{total\,intake} - 3.3 \times DMI)} \qquad \text{EQUATION 6}$$

Where:

C = 3.4 for cows and bulls (assume the same value for steers and heifers)
$W_{total\ intake}$ = L/hd/d

DMI = kg/hd/d

Excreted urine and faeces (manure) water had a manure moisture content of 96%.

Additional Water Use Activities

Additional water use activities consisted of trough cleaning water, evaporation from the troughs, and office and amenities water usage. These data were estimated from data collected at a series of Australian feedlots by Davis et al. (2008a).

Feedlot Pen Water Balance

There were three main outputs from the feedlot pen water balance; evaporation losses as a result of respiration and perspiration, transfers with cattle live weight transported off farm, and flows to the manure management system. Table 51 show the feedlot water supply balance.

TABLE 51 – FEEDLOT WATER SUPPLY BALANCE

Source	Source Description	Use Description	Volume (L / finished animal)	Volume (ML)	Uncertainty (SD or range)
Inputs (source and use)					
Groundwater (stock)	Water sourced from bore	Feedlot water supply (includes drinking water, losses, cleaning, maintenance)	508.7	61.5	1.1
Direct capture from rainfall	Direct supply from supply dam		2034.9	246.1	1.1
Feed (feed moisture and metabolic water)		Water taken up by plants, assumed to be green water source	384.1	46.5	1.43
Cattle (purchased animals brought to the farm)		Water accounted for in grazing processes	125.6	15.2	1.43
Total inputs			3053.4	369.3	
Outputs (source and use)					
Groundwater (stock)	Drinking water lost via the physiological processes of perspiration and respiration	Evaporative use	423.7	51.2	1.43
	Drinking water assimilated into the animal	Catchment transfer	157.4	19.0	1.43

	product				
	Drinking water excreted in manure and urine	Evaporative use	2328.4	281.6	1.43
	Minor uses	Evaporative use	145.7	17.6	1.1
Total outputs			3055.3	369.5	
Balance			-1.9	-0.2	

Feedlot Site Water Balance

Australian feedlots are constructed to control leaching and runoff of water within a controlled drainage area to minimise environmental impacts from eutrophication, under strict design and operational controls. One consequence of this is that runoff from the feedlot is greatly increased, because of the large area of compacted pen surface and roads. This water is contained in catchment dams (>100 ML). Because of the climatic conditions experienced in Australia, such dams regularly operate with a negative water balance (evaporation is greater than inflow) and therefore no water leaves the system. Only in years with exceptionally high rainfall will these systems release any water. Because of the significantly altered hydrological conditions, it was more appropriate to assess water 'use' from the feedlot site by comparing the total annual runoff leaving the site with a reference site (i.e. the site in the absence of the feedlot). This broadly followed the approach recommended by Mila I Canals et al. (2009).

To achieve this, runoff from the reference site (the total feedlot controlled drainage footprint) was determined using USDA-SCS KII curve numbers (USDA-SCS 1972, USDA NRCS 2007). The difference between water released from the feedlot to the natural environment (zero) and the volume of runoff water leaving the reference site was attributed to the feedlot as water use. Table 52 shows the volume of runoff water from reference site attributed to feedlot cattle production.

TABLE 52 – FEEDLOT RUNOFF CAPTURE VOLUME FOR FEEDLOT

	Units	Feedlot
Runoff from reference land occupation	ML/yr	14.52
Runoff from feedlot controlled drainage area	ML / yr	0.0
Consumptive water use attributed to cattle production	L / finished animal	108

Appendix 4 – Herd Dynamics and Modelling GHG Emissions

Grazing System Herd Dynamics

A detailed understanding of herd dynamics was required for several important inventory processes, particularly the estimation of direct GHG emissions and water use. Both farms had very good records for important productivity parameters such as weaning percentage, age and weight at weaning, and age / weight of sale cattle. Some growth rate data were also available for growing cattle. Both farms were able to supply some data for sale cattle that included estimated age (dentition) and slaughter weight. These data were obtained over a minimum of three years and in the case of the NE QLD farm, six years. From these data, cattle could be traced from birth to sale, allowing accurate estimation of mean sale age. This was critical to determining accurate growth rates.

Weight for age data for backgrounding cattle were extrapolated from data collected at weaning and sale. Growth rate was assumed to decline slightly with age and was strongly influenced by season. Because data were not available to validate the weight for age of backgrounding cattle transferred to the finishing system, a small degree of error may exist when viewing these data independently, or where alternative finishing scenarios were modelled.

For each sub-system a liveweight balance was established. This followed a simple formula:

Total liveweight gain (kg) = total liveweight in – total liveweight out

This accounted for loss in mortalities. Neither farm recorded the exact date of mortalities, though more are known to occur at certain times (such as calving). This was important to determine, because impacts accrued over the year for animals that subsequently die represent an additional burden on the herd. In this study a conservative approach was taken, where all mortalities were assumed to occur late in the year, meaning that feed, water and emissions associated with these animals were included in the whole herd impacts.

Grazing System Enteric Methane

Enteric methane estimation is typically based on DMI or gross energy intake. Kennedy and Charmley (2012) reported on a study of 13 Brahman cattle fed 22 diets from combinations of five tropical grass species and five legumes. This study predicted that methane yields for the Australian tropical beef herd were 19.6 g/kg forage dry matter intake. This represents a large downward revision of the methane emissions that can be attributed to the northern Australian beef herd grazing tropical pastures.

In order to calculate feed dry matter intake (DMI) from live weight and live weight gain the following equation is used:

$$\boldsymbol{DMI = (1.185 + 0.00454W - 0.0000026W^2 + 0.315LWG)^2 \times MA} \qquad \text{Equation 7}$$

Where:

W = live weight in kg
LWG = live weight gain in kg/head/day

It is usual for feed intake to increase considerably when lactating occurs. The additional feed intake required during milk production is given by the equation:

$$\boldsymbol{MA = (LC \times FA) + ((1 - LC) \times 1)} \qquad \text{EQUATION 8}$$

Where:

LC =proportion of cows>2 years old that are lactating
FA =feed adjustment (varies between 0 and 1.3 – see Table 6.B.5 (DCCEE 2010))

Based on the study by Kennedy and Charmley (2012), the following regression equation was used to predict the enteric methane emissions from the cattle grazing on tropical pastures in Queensland:

$$\boldsymbol{CH_4\ yield = 19.6 \times DMI} \qquad \text{EQUATION 9}$$

Grazing System Manure Emissions

Manure Methane Emissions

The DCCEE (2010) report that methane emissions from pasture fed cattle manure using the equation developed by Gonzalez-Avalos and Ruiz-Suarez (2001).

$$\boldsymbol{M = I \times (1 - DMD) \times MEF} \qquad \text{EQUATION 10}$$

Where:

M = methane yield (kg CH_4/head/day)
I = feed intake (kg dry matter/head/day) DMD = dry matter digestibility (%)
MEF = manure emission factor (0.000014 for temperate regions, and 0.000054 for tropical regions)

Manure Nitrous Oxide Emissions

In order to calculate the nitrous oxide emissions from pasture fed cattle, it is first necessary to determine the nitrogen content of the excreted faeces and urine to pasture. This is found by calculating the crude protein content (CPI) and amount of nitrogen retained by the body (NR).

The crude protein intake CPI (kg/head/day) of beef cattle is calculated using:

$$\boldsymbol{CPI = I \times CP + (0.032 \times MC)} \qquad \text{EQUATION 11}$$

Where:
I = dry matter intake (kg/head/day)
CP = crude protein content of feed dry matter expressed as a fraction
MC = milk intake (kg/head/day).

Nitrogen excreted in faeces (F kg/head/day) is found using a similar method to that for feedlot, however the contribution from milk protein is included in this case:

$$\boldsymbol{F} = \left\{\mathbf{0.3}\left(\boldsymbol{CPI} \times \left(\mathbf{1} - \left[\frac{(\boldsymbol{DMD}+\mathbf{10})}{\mathbf{100}}\right]\right)\right) + \mathbf{0.105}(\boldsymbol{ME} \times \boldsymbol{I} \times \mathbf{0.008}) + \mathbf{0.08}(\mathbf{0.032} \times \boldsymbol{MC}) + (\mathbf{0.0152} \times \boldsymbol{I})\right\} / \mathbf{6.25}$$ EQUATION 12

Where:
DMD = dry matter digestibility (expressed as a %)
ME = metabolise energy (MJ/kg DM)
I = feed intake (kg DM/head/day)
MC = milk intake (kg/head/day)

Table 53 shows the crude protein content of the dry matter fraction of pasture assumed for this study. Where site specific data or better estimates were available these were substituted.

TABLE 53 – DRY MATTER CRUDE PROTEIN (CP) CONTENT OF PASTURE

Farm	CP (%)
NE QLD	9.8
SW QLD	9.8

The quantity of nitrogen that is retained within the body (NR kg/head/day) is determined as the amount of nitrogen retained as body tissue and milk:

$$\boldsymbol{NR} = \left\{(\mathbf{0.032} \times \boldsymbol{MP}) + \left\{\mathbf{0.212} - \mathbf{0.008}(\boldsymbol{L}-\mathbf{2}) - \left[\left(\mathbf{0.140} - \mathbf{0.008}(\boldsymbol{L}-\mathbf{2})\right) / \left(\mathbf{1} + \boldsymbol{exp}\left(-\mathbf{6}(\boldsymbol{Z} - \mathbf{0.4})\right)\right)\right]\right\} \times (\boldsymbol{LWG} \times \mathbf{0.92})\right\} / \mathbf{6.25}$$ EQUATION 13

Where:
MP = milk production (kg/head/day)
L = relative intake
Z = relative size (liveweight/standard reference weight)
LWG = liveweight gain (kg/day)

The amount of nitrogen excreted in urine (U) is found using the equation:

$$\boldsymbol{U} = \left(\frac{\boldsymbol{CPI}}{\mathbf{6.25}}\right) - \boldsymbol{NR} - \boldsymbol{F} - \left[\frac{(\mathbf{1.1} \times \mathbf{10^{-4}} \times \boldsymbol{LW}^{\mathbf{0.75}})}{\mathbf{6.25}}\right]$$ EQUATION 14

Where:
LW = average seasonal liveweight of animal

The nitrous oxide emissions from faecal and urinary nitrogen voided onto pasture are calculated using:

$$\boldsymbol{N_2O\ emissions} = (\boldsymbol{F} + \boldsymbol{U}) \times \boldsymbol{MMS} \times \boldsymbol{EF}_{(\boldsymbol{MMS})} \times \boldsymbol{C_g}$$ EQUATION 15

Where:

MMS = the fraction of nitrogen that is voided to pasture – assumed to be 100%.
$EF_{(MMS)}$ = emissions factor (N_2O-N kg/N excreted). This is 0.005 for faeces and 0.004 for urine.
C_g = 44/28 factor to convert elemental mass of N_2O to molecular mass.

Feedlot Herd Dynamics

Livestock data were readily accessible at the feedlot. The data were provided from herd management databases at the feedlot for a 12 month period: Livestock movements, days on feed, mortalities, average daily gain and feed intake. Additionally, digestibility, nitrogen, phosphorus and ash levels in the feed were available from laboratory analyses. These data were used to predict enteric emissions and manure production using a modified version of BeefBal (QPIF 2004). BeefBal estimates the total solids (TS), volatile solids (VS), fixed solids (FS, or ash), nitrogen (N), phosphorus (P), potassium (K) and salt (as sodium chloride) in the manure from a feedlot. The DMDAMP model (van Sliedregt et al. 2000), within BeefBal, is used to calculate TS excreted, while N, P, K, salt and FS excretion is determined using a mass balance (Watts et al. 1994). Volatile solids excretion is determined by difference.

Feedlot Enteric Methane

Enteric methane was modelled using the DCCEE (2010) methodology for feedlot cattle, which is based on Moe and Tyrrell (1979). This approach requires the estimation of gross energy intake and then calculates the proportion of this energy that is converted into methane based on the digestibility at maintenance of the feed energy and the level of feed intake relative to that required for maintenance. The equations for methane emission require some detail regarding dietary components, specifically, the proportion of soluble residue, hemicellulose and cellulose in the diet.

The formula for enteric methane yield (Y– MJ CH_4/head/day) is as follows:

$$\boldsymbol{Y = 3.406 + 0.510SR + 1.736H + 2.648C} \qquad \text{Equation 16}$$

Where:

SR	=	intake of soluble residue (kg/day)
H	=	intake of hemicellulose (kg/day)
C	=	intake of cellulose (kg/day)

Each of SR, H and C are calculated from the total intake of the animal, the proportion of the diet of each class of animal that is grass, legume, grain (including molasses) and other concentrates and the soluble residue, hemicellulose and cellulose fractions of each of these components.

Hence:

$$\boldsymbol{SR} = \left(\boldsymbol{I} \times \boldsymbol{P_{grain}} \times \boldsymbol{SR_{grain}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{conc}} \times \boldsymbol{SR_{conc}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{grass}} \times \boldsymbol{SR_{grass}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{legume}} \times \boldsymbol{SR_{legume}}\right)$$

EQUATION 17

$$\boldsymbol{H} = \left(\boldsymbol{I} \times \boldsymbol{P_{grain}} \times \boldsymbol{H_{grain}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{conc}} \times \boldsymbol{H_{conc}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{grass}} \times \boldsymbol{H_{grass}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{legume}} \times \boldsymbol{H_{legume}}\right)$$

EQUATION 18

$$\boldsymbol{C} = \left(\boldsymbol{I} \times \boldsymbol{P_{grain}} \times \boldsymbol{C_{grain}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{conc}} \times \boldsymbol{C_{conc}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{grass}} \times \boldsymbol{C_{grass}}\right) + \left(\boldsymbol{I} \times \boldsymbol{P_{legume}} \times \boldsymbol{C_{legume}}\right)$$

EQUATION 19

Where:

I = intake (kg/day)
Pgrain = proportion of grains in feed
Pconc = proportion of concentrates in feed
Pgrass = proportion of grasses in feed
Plegume = proportion of legumes in feed
SR, H or C grain = soluble residue, hemicellulose or cellulose content of grain
SR, H or C conc = soluble residue, hemicellulose or cellulose content of other concentrates
SR, H or C grass = soluble residue, hemicellulose or cellulose content of grasses
SR, H or C legume = soluble residue, hemicellulose or cellulose content of legumes

The total daily production of methane, M_{ij} (kg CH_4/head/day) is thus:

$$\boldsymbol{M} = \boldsymbol{Y}/\boldsymbol{F}$$

EQUATION 20

Where:

F = 55.22 MJ/kg CH_4

The study by Beauchemin et al. (2010) determined that feeding oil in the feedlot ration, results in a reduction in enteric methane emissions. This reduction is equivalent to 5.6% for every 1% of oil fed in the diet. The feedlot used in this study was assumed to feed oil in the ration and therefore the methane production was reduced. Table 54 show the proportion of oil fed in the diet along with reduced daily enteric methane production.

The DCCEE provide default values for daily feed intake and feed properties for Australian feedlot cattle. However, for the feedlot under investigation, actual data were available and were substituted into the equations described previously. Key differences between the DCCEE default assumptions and the actual data collected from the feedlot relate to daily dry matter intake (DMI) and the proportion of grain, grass, legume and concentrate in the diets. Table 54 shows the daily feed intake and feed properties for the feedlot used in this study.

TABLE 54 – DAILY FEED INTAKE AND FEED PROPERTIES FOR FEEDLOT

		DCCEE (2010)	Actual data
Daily Intake (assume DMI)	(kg/day)	8.9	8.9
Proportion of grains in feed	(%)	77.9	84.1
Proportion of concentrates in	(%)	4.8	6.1

feed			
Proportion of grasses in feed[1]	(%)	13.8	9.8
Proportion of legumes in feed	(%)	3.5	0.0
Proportion of oil in feed	(%)	n.a	3.7
Enteric methane production – without accounting for oil	(kg/hd/yr)	0.18	0.17
Enteric methane production – oil accounted for	(kg/hd/yr)	n.a	0.14

[1] forage hay / silage classified under grasses

Feedlot Manure Emissions

Greenhouse gas emission estimation from manure management relies on the prediction of specific manure properties; excreted volatile solids (VS) and nitrogen (N). Other nutrient components of manure are also relevant for estimating nutrient by-product value in manure.

The mass balance approach is recommended by the IPCC (Dong et al. 2006) as the state of the art for estimation of manure losses from intensive livestock. The BEEFBAL program enables the estimation of excreted VS and traces these through the feedlot system with a series of partitioning and emission estimates. VS is calculated using the dry matter digestibility of the diet as per DCCEE (2010). The program accounts for partitioning between the effluent pond and solid storage, and traces VS through to land application as effluent or manure.

BEEFBAL is a more comprehensive basis for estimating GHG from the whole manure management system at the feedlot than the DCCEE method. However, the program requires expert user input to specify several important partitioning factors. In the present study, these were determined from industry experts with extensive knowledge of feedlot manure and effluent treatment systems. Figure 16 shows a simplified mass balance for VS at Australian feedlots.

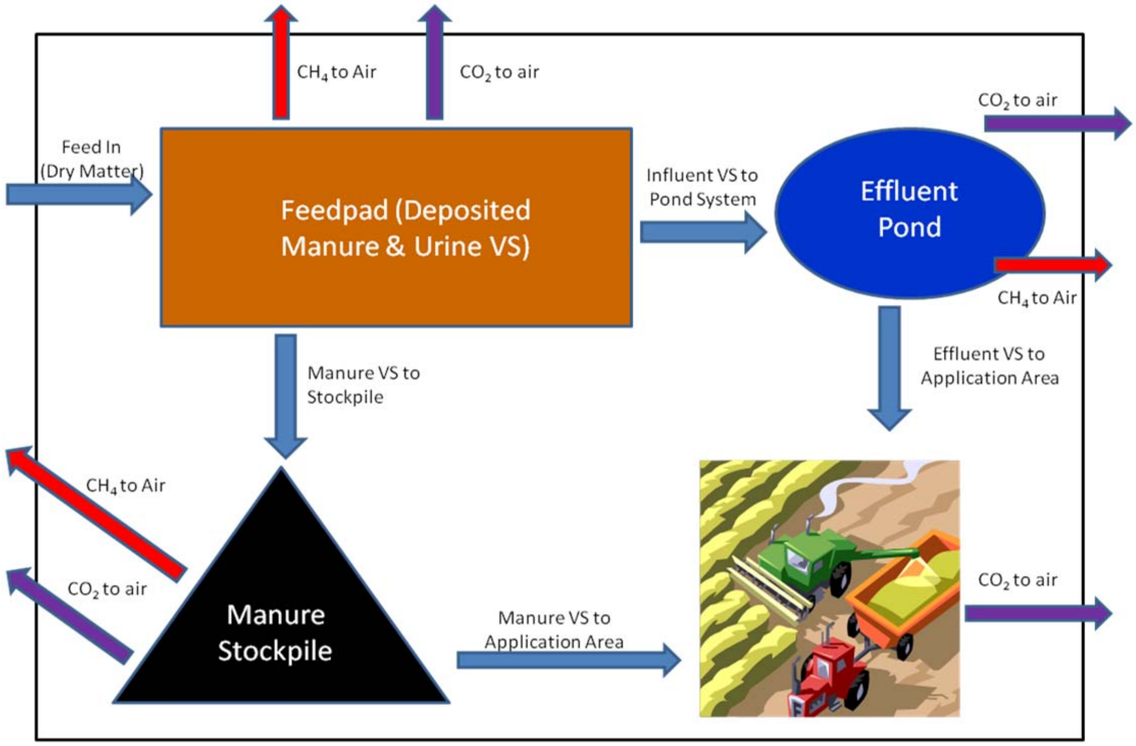

FIGURE 16 – THEORETICAL MASS BALANCE FOR EXCRETED VOLATILE SOLIDS IN AUSTRALIAN FEEDLOTS

The methane emission factors and ranges used for this study are summarised in Table 55.

TABLE 55 – FEEDLOT METHANE EMISSION FACTORS USED IN THIS STUDY

Emission source	Best Science	Range	Reference for emission factor used in the theoretical mass balance
Feedpad (CH_4)	5 % or 1.5%[1]	2.5-7.5%[a] 0.75-2.25%[b]	DCCEE 2010 – feedlot beef
Stockpile (CH_4)	5 %	4.5-5.5%	IPCC 2006 default
Effluent Pond (CH_4)	80 %	64-88%	DCCEE 2010 – dairy industry

[1] feedpad methane emission factor varies
[a] range for MCF of 5%
[b] range for MCF of 1.5%

Manure Nitrous Oxide Emissions

The majority of nitrogen consumed by feedlot cattle as protein in the diet is excreted in manure and urine. Excreted nitrogen is rapidly lost to the atmosphere through a number of pathways. Of these, direct nitrous oxide emissions contribute directly to the GHG profile of the feedlot. Additionally, emissions of ammonia contribute to indirect GHG emissions when ammonia is deposited to surrounding land and re-emitted as nitrous oxide. Hence, both direct nitrous oxide emissions and ammonia emissions are important for the estimation of total GHG.

Estimation of nitrogen emissions begins with calculation of the total mass of nitrogen excreted from the cattle. Excretion is determined by difference from estimating crude protein intake and retention within the animal. Crude protein in the feedlot ration was 18.3%.

Feedpad Emissions

The total emissions of nitrous oxide from the feedpad (designated 'Drylot' by the DCCEE) are calculated as follows:

$$\boldsymbol{Faecal_{MMS} = AF \times MMS \times EF_{(MMS)} \times C_g} \qquad \text{Equation 21}$$

$$\boldsymbol{Urine_{MMS} = AU \times MMS \times EF_{(MMS)} \times C_g} \qquad \text{Equation 22}$$

$$\boldsymbol{Total_{MMS} = Faecal_{MMS} + Urine_{MMS}} \qquad \text{Equation 23}$$

Where:

MMS = the fraction of the annual nitrogen excreted (AU + AF) that is managed in the different manure management systems.

$EF_{(MMS)}$ = emissions factor for the different manure management systems.

C_g = 44/28 factor to convert elemental mass of N_2O to molecular mass

WE applied an updated pad nitrous oxide emission factor of 0.005 kg N_2O-N / kg N excreted, based on Australian research collated by Muir (2011). This is lower than the emission factor recommended by the DCCEE (2010).

Following excretion from the feedpad, there is a partitioning of nitrogen between solid and liquid (effluent) storage. This results in emission losses from both solid and liquid storage. Figure 17 shows a generalised theoretical mass balance of nitrogen at a feedlot.

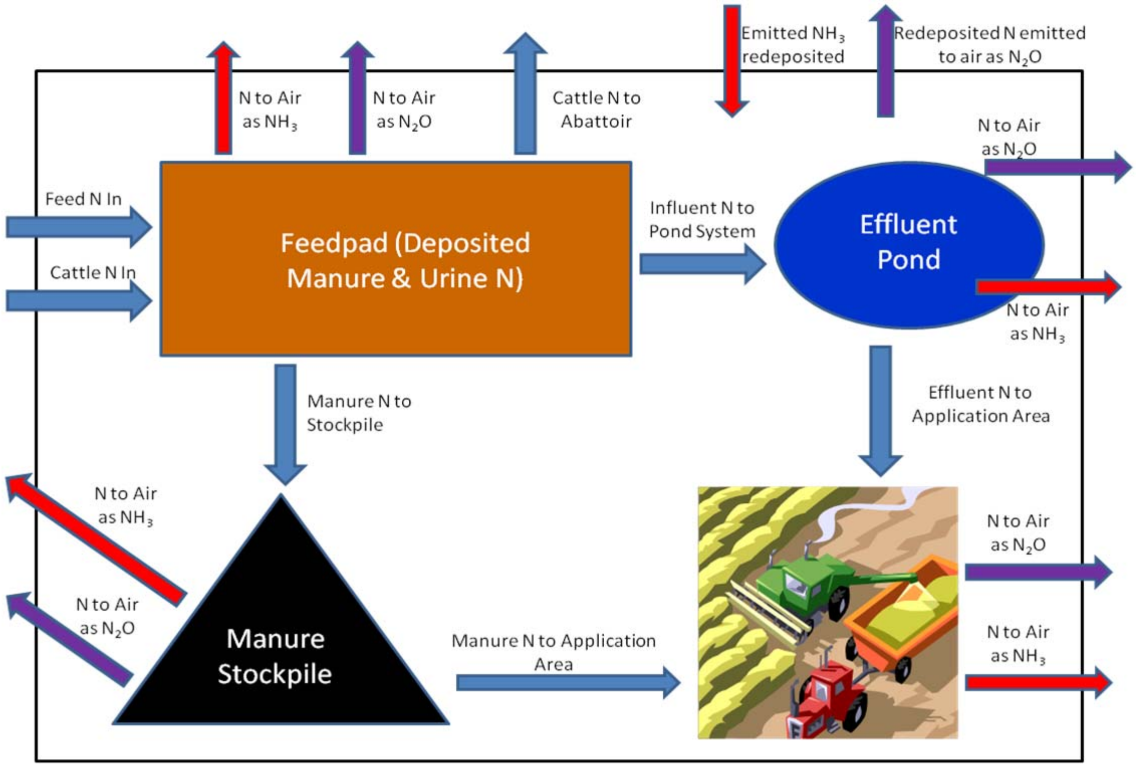

FIGURE 17 – THEORETICAL MASS BALANCE FOR EXCRETED NITROGEN IN AUSTRALIAN FEEDLOTS

Table 56 summarises the nitrous oxide and ammonia emission factors for feedlots used for the two approaches described in this study.

TABLE 56 –FEEDLOT NITROUS OXIDE AND AMMONIA EMISSION FACTORS USED IN THIS STUDY

Emission source	Factor	Reference for emission factor used in the theoretical mass balance
Storage and Feedpad (N_2O)	0.5 %	Muir (2011)
Feedpad (NH_3)	75 %	Watts et al. (2011)
Manure Storage (N_2O)	0.5%	IPCC 2006 default
Manure Storage (NH_3)	25 %	Watts et al. (2011)
Effluent Pond (N_2O)	0.1%	DCCEE (2010) – dairy industry
Effluent Pond (NH_3)	35 %	Watts et al. (2011)
Manure Application (N_2O)	1 %	DCCEE (2010) – manure application
Manure Application (NH_3)	20 %	Watts et al. (2011)
Effluent Application (N_2O)	1 %	DCCEE (2010) – dairy industry
Effluent Application (NH_3)	20 %	Watts et al. (2011)
Atmospheric deposition (N_2O)	1 %	DCCEE (2010)

Leaching and runoff nitrogen was not considered as a source from the feedlot controlled drainage area, which was designed to restrict both leaching and runoff via compaction and containment of effluent.

Soil Carbon Flux

The estimation of soil carbon sequestration is contentious. There remains no general agreement on the method of calculation or how to determine the time frame over which change occurs due to the uncertainty over whether the process can be considered to continue in the long-term. Nevertheless, it is recognised that in some cases carbon sequestration may represent a significant quantum of removal of atmospheric CO_2, predominantly by incorporation into soil organic matter, and particularly where management changes from intensive cultivation to perennial pasture or forest cover. There is limited data on which to base an internationally agreed methodology and, based on input from a range of country experts, the most appropriate treatment at present is that in ISO 14067 DIS (2012 draft) whereby the carbon sequestration can be estimated based on use of IPCC (2006), but must not be included in the final aggregated GHG value. It may, however, be presented separately to indicate the potential effect.

The study by Dalal and Chan (2001) determined that carbon loss from soil from rain fed cropping systems in the Australian cereal belt can be found using the following equation:

$$\boldsymbol{Soil_{C\ Loss}} = \frac{\mathbf{0.1756 \times 100}}{\boldsymbol{Clay\ content}} \qquad \text{EQUATION 24}$$

It was assumed that soils with 45% clay content were representative of the Queensland cropping zone assumed for this study. The amount of C loss was determined to be 0.39 t ha yr. It was assumed that 25% of soils in this cropping region are losing carbon at this rate, while the remainder are being well managed with zero tillage and no soil C loss. Therefore, the soil C loss was assumed to be 0.0975 t C ha yr.

Summary of GHG Calculation Methods and Factors

The parameters and equations used in this study to determine the GHG emissions from grazing and feedlot beef are summarised in Table 57 and Table 58, along with the assumed uncertainty.

TABLE 57 – KEY GHG PARAMETERS USED FOR GRAZING CATTLE WITH UNCERTAINTY

Emission source	Key parameters / model	Assumed Uncertainty	Reference
Enteric methane (temperate climate)	M(kg/hd) = I (kg DM/hd) x 19.6	± 14%	Kennedy & Charmley (2012)
Manure methane	M (kg/hd) = I (kg DM/hd) x (1 - DMD) x MEF	± 20%	DCCEE (2010)
Manure nitrous oxide	Urinary N – 0.004 kg N_2O-N / kg N in urine. Faecal N – 0.005 kg N_2O-N / kg N in faeces.	± 50%	DCCEE (2010)
Manure ammonia	0.2 kg NH_3-N / kg N of excreted in manure	± 20%	DCCEE (2010)
Indirect nitrous oxide from ammonia losses	0.01 kg N_2O-N / kg NH_3-N volatilised	± 50%	DCCEE (2010)
Indirect nitrous oxide from leaching and runoff	0.0125 kg N_2O-N / kg NO_3-N lost in leaching and runoff	± 50%	DCCEE (2010)

TABLE 58 – KEY GHG PARAMETERS USED FOR FEEDLOT CATTLE WITH UNCERTAINTY

Emission source	Key parameters / model	Assumed Uncertainty	Reference
Enteric methane	M (kg/hd) = (3.406 + 0.510SR + 1.736H + 2.648C) / F (MJ / kg CH_4)	± 20%	DCCEE (2010) – from Moe and Tyrrell (1979)
Manure methane	M (kg/hd) = VS (kg/head) x B_o (0.17 m^3 CH_4/kg VS) x MCF x p (0.622 kg/m^3)	± 20%	DCCEE (2010)
Manure nitrous oxide	Urinary N – 0.005 kg N_2O-N / kg N in urine. Faecal N – 0.005 kg N_2O-N / kg N in faeces.	± 50%	Muir (2011)
Manure ammonia	0.75 kg NH_3-N / kg N of excreted in manure	± 20%	Watts et al. (2011)
Indirect nitrous oxide from ammonia losses	0.01 kg N_2O-N / kg NH_3-N volatilised	± 50%	DCCEE (2010)

Appendix 5 – Meat Processing Inventory Methods

Foreground data

Data for all unit operations within the system boundary was collected. Wherever possible, data for the specific operations was collected. However this was not been possible for all processes, in which case best-available representative data from literature or industry consultation was used. The following summarises the types of data collected. Details of the data collected, and their sources are provided in the following sections for the domestic and export supply chains respectively.

Operation-specific data was collected for:

- An abattoir producing beef for domestic consumption; and
- An abattoir producing beef for export.

Combinations of best-available representative data from literature and educated estimates were used for:

- secondary processing operations in Japan;
- warehousing operations in Australia and Japan;
- retail operations in Australian and Japanese supermarkets;
- meat wastage and disposal routes for meat waste;
- transport modes and distances for overseas shipping, distribution and consumer transport; and
- consumer behaviour in relation to the storage and consumption of beef in the home.

Data for the following aspects could not be included:

- paunch waste management, as the disposal route could not be determined;
- processes for the production of finished packaging products. The production of primary packaging materials (plastic pellets, cardboard) has been included, but not the downstream processes of producing finished packaging products (plastic film, plastic trays, polystyrene trays, cardboard boxes etc.).
- truck washing and maintenance associated with the transport fleets used for product distribution. Only truck operation has been included.

Background data included

For the following background processes occurring in Australia, data from the Australian Life Cycle Inventory database has been used (Life Cycle Strategies 2007) :

- electricity (based on average supply mix of low voltage electricity for Queensland)
- energy from coal (based on combustion of Queensland thermal coal)

- energy from natural gas (based combustion of average supply mix of high pressure mains gas for Australia)
- water (based on reticulated drinking water in Brisbane)
- cleaning chemicals (based on caustic soda)
- primary packaging materials (corrugated board and low density polyethylene film)
- passenger transport (car, train and bus operations for Australia)
- truck transport for distribution (based on rigid truck operation)
- wastewater treatment (based on sewage treatment in Brisbane)
- landfilling of meat waste (based on average landfilling of food, other organics and inert wastes for Australia).

For the following processes occurring in Japan, Japanese data from the Ecoinvent database (Swiss Centre for Life Cycle Inventories 2009) has been used:

- shipping (based on transoceanic freight ship)
- electricity (based on average supply mix of low voltage electricity for Japan)
- energy from natural gas (based combustion of average supply mix of mains gas for Japan)

For other processes occurring in Japan, for which Japan-specific data wasn't available, the following data has been used:

- truck transport for distribution (based on rigid truck operation for Australia)
- passenger transport (train and bus operations for Europe, and car operations for Australia)
- landfilling of meat waste (based on average landfilling of food, other organics and inert wastes for Australia).
- primary packaging materials (corrugated board and low density polyethylene film)

Detailed Inventory Assumptions – NE Supply Chain

Abattoir processes in the export supply chain, producing beef for export consumption:

- Data are based on processes occurring at an export abattoir in Southern QLD, which is assumed to be representative of an export abattoir in Townsville.
- Data used are averages for the year 2010, provided by abattoir staff in January, 2011
- Beef cattle only are processed at the abattoir.
- Final beef products from the abattoir are frozen (-18^{o}C) and chilled (0^{o}C) prime cuts of beef in boxes (approx 20-30kg per box).
- Co-products from the abattoir are edible offal, hides and rendering material. Rendering materials (fat, meat scraps, bone, blood) are further processed in an on-site rendering plant into rendered co-products (pet food, meat meal, tallow and dry blood).

- Electricity is sourced from the grid; 84% is used in the abattoir and 16% is used in the rendering plant
- Thermal energy (steam and hot water) is generated on site in a natural gas and coal-fired boiler; 5% of steam is consumed in the abattoir and 95% is used in rendering
- The plant has anaerobic treatment ponds and therefore produces methane during treatment.
- Wastewater from the abattoir and rendering plant is treated on site using uncovered anaerobic treatment ponds (releasing methane, ammonia and nitrous oxide), and then discharged directly to a local waterway under an Environmental Authority
- Solid waste (paunch manure, sludge and boiler ash) are assumed to be disposed to landfill.

Overseas shipping:

- In the absence of operation-specific information, it was assumed that boxed, primal cuts of beef from the abattoir in Townsville are first transported 16km, in rigid diesel trucks, to a refrigerated warehouse at the Port of Townsville. They are then assumed to be shipped 6461km, by ocean freighter, to the Port of Yokohama, Japan, where they are again stored a refrigerated warehouse. They are then assumed to be transport 20km, by rigid diesel truck, to secondary processing centres. For reference, the greenhouse gas impacts from shipping were 0.0107 kg CO_2-e / t.km.
- In the absence of operation-specific information about warehousing in Australia and Japan, general data related to warehousing from literature was used (DEFRA 2008).

Secondary Processing (in Japan):

- In the absence of operation-specific information about the secondary processing of primal beef cuts in Japan, data was generated by scaling down the operational data for Australian abattoir operations.

Distribution from secondary processors to retail outlets

- In the absence of operation-specific information, it was assumed that beef products from secondary processing are distributed, in rigid diesel trucks, directly to supermarkets around Japan. 30% is assumed to be distributed 50km, 45% is distributed 500km and 25% is distributed 1200km.

Retail

- Beef products are assumed to be retailed to Japanese consumers through a range of supermarkets, with an average-sized store of 1700 m^2 and a turnover of $A16.4 million/yr assumed to be representative (Brodribb et al. 2009, JSA 2010, Younger 1995)
- In the absence of specific information about supermarket operations in Japan, generic data for this sized store from literature sources were used (Younger 1995).
- For the purpose of allocating the impacts of supermarket operations (not including refrigeration) to beef products:

  - 1% of the total mass of all products in the supermarket is beef, assuming 30% of all products are refrigerated, 25% of which is meat, and 20% of which is beef (estimate).
  - 3% of the total economic value of all products in the supermarket was assumed to be beef products (estimate);
- For the purpose of allocating the impacts of supermarket refrigeration to beef products:
  - 3% of the mass of all refrigerated product in the supermarket is beef, assuming 25% (by mass) of refrigerated product is meat, and 20% (by mass) of this is beef (estimate).
  - 8% of the economic value of all refrigerated product in the supermarket is beef, assuming 53% (by value) of refrigerated product is meat (estimate based on (Lunde & Feitz 2004), and 25% (estimate) of this is beef (estimate);

Transport to the home:

- Beef purchases per household (2.55 people/hh) (JSB 2010) were assumed to be 0.45 kg/hh/week.
- Beef products were assumed to be acquired as part of grocery shopping trips, assuming 2.5 trips/hh/wk (JMI 2009)
- The transport distance between supermarket and the home was assumed to be 10km (round trip), and the modal split was assumed to be 20% car, 15% train, 15% bus, and 50% walking (estimate).
- For the purpose of allocating the impacts of grocery transportation to beef products:
  - the total mass of groceries transported was assumed to be 22.4 kg/grocery trip (JMI 2009), based on 2% being beef product (estimate);
  - the total economic value of groceries transported was assumed to be $84/grocery trip (JSB 2010), 4% of which is beef product (estimate).

Home storage:

- Beef products were assumed to be stored in home refrigerator / freezer; 20% in the freezer compartment for 7 days, and 80% in the fridge compartment for 3 days.
- Impacts of operating home fridge / freezers were based on a 400-450L, vertical fridge-freezer, with a 4-star energy rating, using 432 kWh/yr (Motoshita 2010).
- For the purpose of allocating the impacts of home refrigeration to beef products:
  - the total mass of groceries stored in the fridge/freezer was assumed to be 20kg (estimate), 2% of which is beef product (estimate);
  - the proportion of the total economic value of groceries stored in the fridge/freezer that is beef was assumed to be 4% (estimate).

Consumption (cooking and wash-up):

- Beef products are assumed to be cooked using a mixture of gas (90%) and electricity (10%) (Panasonic 2010a)

  - Impacts of cooking beef products with gas were based on a 5 MJ/hr cook-top, requiring 1 min to cook a 250g steak.
  - Impacts of cooking beef products with electricity were based on a 2.7kW cook-top, requiring 1 min to cook a 250g steak.
- Eating implements were assumed to be washed in a dishwasher (35%) or by hand (65%) (Panasonic 2010b).
  - Impacts of dishwasher operation were based on a dishwasher using 200kWh/yr of electricity and 10L of water/load (Australian Government 2010, DCCEE 2011), assuming one load per day.
  - Impacts of hand washing were based on using 18L of water total per wash; 50% cold water, 50% electricity-heated water.
- For the purpose of allocating the impacts of dishwashing to beef products, the same allocation factors as applied to the transport of groceries to the home were used.

Wastage of beef product along the supply chain:

- Wastage of beef product at the abattoir and secondary processing is assumed to be negligible as it would be sent to rendering for processing into lower-value co-products.
- Wastage of beef during retail assumed to be 5%
- Wastage of beef during distribution and home transport is assumed to be negligible.
- Wastage of beef in home storage, preparation, cooking and consumption assumed to be 5%
- All wasted beef is assumed to be disposed to a managed landfill, at which around half of the methane emitted is capture for electricity generation.

Detailed Inventory Assumptions – SW Supply Chain

Abattoir process in the domestic supply chain, producing beef for domestic consumption:

- Data are 3 year averages (mid 2007- mid 2010), provided by abattoir staff in Nov, 2010
- Beef cattle only are processed at the abattoir, not weaners.
- Co-products from the abattoir are edible offal, hides and rendering material. Rendering materials (fat, meat scraps, bone, blood) are further processed in an on-site rendering plant into rendered co-products (pet food, meat meal, tallow and dry blood).
- Electricity is sourced from the grid; 93% is used in the abattoir and 7% is used in the rendering plant
- Thermal energy (steam and hot water) is generated on site in a coal-fired boiler; 20% of total steam is used in the abattoir and 80% is used in the rendering plant
- Wastewater from the abattoir and rendering plant is treated on site using aerobic digestion (therefore does not produce methane) and then discharged to sewer as trade waste.
- All solid waste (paunch manure, sludge and boiler ash) is assumed to be disposed to landfill.

Distribution from abattoir to retail outlet:

- In the absence of operation-specific information about distribution, it was assumed that beef products from the abattoir are distributed in rigid diesel trucks, via a chilled distribution centre to supermarkets in SE Queensland and Northern NSW. 40% is assumed to be distributed within Brisbane (50km), 40% to regional centres within 300km radius, and 20% to NSW (400km).
- In the absence of operation-specific information about the chilled distribution centre, data related to warehousing from literature was used (DEFRA 2008).

Retail:

- Beef products are assumed to be retailed to SE Queensland and Northern NSW consumers through supermarkets, with a large-size store of 3,800 m^2 and a turnover of $A34 million/yr assumed to be representative.
- In the absence of specific information about supermarket operations in SE Qld and Northern NSW, generic data for this sized store from literature sources were used (Brodribb et al. 2009, Younger 1995).
- For the purpose of allocating the impacts of supermarket operations (not including refrigeration) to beef products:
    - 2% of the total mass of all products in the supermarket is beef products, assuming 30% of all product is refrigerated, 25% of which is meat, and 25% of which is beef (estimate).
    - 5% of the total economic value of all products in the supermarket was assumed to be beef products (estimate);
- For the purpose of allocating the impacts of supermarket refrigeration to beef products:
    - 5% of the mass of all refrigerated product in the supermarket is beef, assuming 25% (by mass) of refrigerated product is meat, and 25% (by mass) of this is beef (estimate).
    - 13% of the economic value of all refrigerated product in the supermarket is beef, assuming 53% (by value) of refrigerated product is meat (Lunde & Feitz 2004), and 25% (by value) of this is beef (estimate);

Transport to the home:

- Beef purchases per household (2.6 people/hh - ABS 2010) were assumed to be 0.94 kg/hh/week (ABARE 2009b, MLA 2007).
- Beef products were assumed to be acquired as part of grocery shopping trips, assuming 2.1 trips/hh/wk (The Australia Institute 2010).
- The transport distance between supermarket and the home was assumed to be 9.6km (round trip), and the modal split was assumed to be 85% car, 5% train, 5% bus, and 5% walking (The Australia Institute 2010).
- For the purpose of allocating the impacts of grocery transportation to beef products:
    - the total mass of groceries transported was assumed to be 23.5 kg/grocery trip (ABARE 2009b, MLA 2007), 4% of which is beef product (estimate);

- the total economic value of groceries transported was assumed to be $153/grocery trip of which 13% of this is meat (ABS 2004). 5% of the meat product is assumed to be beef (estimate).

Home storage:

- Beef products were assumed to be stored in home refrigerator / freezer; 40% in the freezer compartment for 14 days, and 60% in the fridge compartment for 3 days.
- Impacts of operating home fridge / freezers were based on a 560L, vertical fridge-freezer, with a 4-start energy rating, using 600 kWh/yr (DCCEE 2011).
- For the purpose of allocating the impacts of home refrigeration to beef products:
  - the total mass of groceries stored in the fridge/freezer was assumed to be 18 kg (estimate), 5% of which is beef product (estimate);
  - the proportion of the total economic value of groceries stored in the fridge/freezer that is beef was assumed to be10% (estimate).

Consumption (cooking and wash-up):

- Beef products are assumed to be cooked using a mixture of gas (44%) and electricity (56%) (ABS 2010).
  - Impacts of cooking beef products with gas were based on a 9 MJ/hr cook-top, requiring 3 mins to cook a 250g steak.
  - Impacts of cooking beef products with electricity were based on a 2kW cook-top, requiring 3 mins to cook a 250g steak.
- Eating implements were assumed to be washed in a dishwasher (35%) or by hand (65%) (DCCEE 2011).
  - Impacts of dishwasher operation were based on a dishwasher using 200kWh/yr of electricity and 10L of water/load (Australian Government 2010), assuming one load per day.
  - Impacts of hand washing were based on using 18L of water total per wash; 50% cold water, 25% gas-heated water, and 25% electricity-heated water.
- For the purpose of allocating the impacts of dishwashing to beef products, the same allocation factors as applied to the transport of groceries to the home were used.

Wastage rates of beef product along the supply chain:

- Wastage of beef at the abattoir was assumed to be negligible as it would be sent to rendering for processing into lower-value co-products.
- Wastage of beef during retail was assumed to be 4.3% (Buzby et al. 2009)
- Wastage of beef during distribution and home transport was assumed to be negligible.
- Wastage of beef in home storage and consumption was assumed to be 10%
- All wasted beef is assumed to be disposed to a managed landfill, at which around half of the methane emitted is capture for electricity generation.