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Life Cycle Assessment of four southern beef supply chains

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Abstract

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 four southern Australian (New South Wales and Victorian) beef supply chains producing beef for several grass and grain fed markets. The study was based on data from four farms and three feedlots producing cattle for a number of grass-fed and grain-fed markets, and includes a comparison of grass, grass/forage and grain finishing systems. Results were presented “per kilogram of live weight” at the farm gate, while results for the backgrounding/finishing systems were compared “per kilogram of live weight gain (LWG)” excluding impacts from breeding. The study covered the following resource use and environmental impact indicators: Cumulative Energy Demand, Consumptive Water Use, Stress Weighted Water Use, Land Occupation, Eutrophication Potential, Phosphorus Flux Potential, Soil Carbon Flux Potential and Total Greenhouse Gas Emissions. This is the most comprehensive study of its type for southern Australian beef production to date.

The results from this study were broadly similar to previous Australian beef LCA research. Compared to the international literature, the results were lower in energy use and similar to, or lower, GHG emissions intensity. Energy demand for the mid weight steers in the study ranged from 4.3-8.1 MJ / kg LW and water use ranged from 107-298 L / kg LW. Stress weighted water use was lower on average, ranging from 23-43 L H2O-e / kg LW for cattle from three of the mid-weight supply chains, and 287 L H2O-e / kg LW from the remaining farm.

In addition to details on the other resource use and environmental impact indicators assessed, recommendations are provided for future research, which could include investigation of a series of scenarios to reduce resource use and emissions intensity from grain fed beef.

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 four southern Australian (NSW and Victorian) beef supply chains producing beef for several grass and grain fed markets. The study was based on data from four farms and three feedlots producing cattle for a number of grass-fed and grain-fed markets, and includes a comparison of grass, grass/forage and grain finishing systems. Results were presented "per kilogram of live weight" at the farm gate, while results for the backgrounding/finishing systems were compared "per kilogram of live weight gain (LWG)" excluding impacts from breeding. The study covered the following resource use and environmental impact indicators: Cumulative Energy Demand, Consumptive Water Use, Stress Weighted Water Use, Land Occupation, Eutrophication Potential, Phosphorus Flux Potential, Soil Carbon Flux Potential and Total Greenhouse Gas Emissions. This is the most comprehensive study of its type for southern Australian beef production to date.

This project demonstrated an association between production efficiency and environmental performance. With some exceptions, higher production efficiency led to better environmental outcomes from the system. This effect could be seen between the market types and finishing systems used. All things being equal, lower impacts were observed from heavier, faster growing slaughter cattle compared to lighter cattle. Even where compensatory (or trade-off) effects were taken into account, improved productivity still tended to result in lower impacts. This trend was driven by improved herd efficiency (kg beef / kg DMI across the whole herd) and was apparent while growth rates were high. Where growth rates declined, this trend was rapidly reversed as the maintenance requirements of the growing cattle increased with age. The feedlot industry plays an important role by improving supply chain productivity through maximising growth rates and slaughter weight, thereby reducing greenhouse gas (GHG) emissions. This noted, feedlot production did require higher levels of energy and cultivated land resources to achieve this. Trends towards higher slaughter weights while maintaining high growth rates in the domestic market should be welcomed as a move towards more efficient, more sustainable beef production.

The results from this study were broadly similar to previous Australian beef LCA research. Compared to the international literature, the results showed lower energy use and similar to, or lower, GHG emissions intensity. Energy demand for the mid weight steers in this study ranged from 4.3-8.1 MJ / kg LW and water use ranged from 107-298 L / kg LW. Stress weighted water use was lower on average, ranging from 23-43 L H_2O-e / kg LW for cattle from three of the mid-weight supply chains, and 287 L H_2O-e / kg LW from the remaining farm. Stress weighted water use was 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. Water use tended to be slightly higher on average than previously estimated by Australian studies, mainly because predicted evaporation losses from farm dams in the present study were much higher than estimated by other studies (if included at all). Nonetheless, water use was several orders lower than most estimates of 'virtual water' use for beef cattle. Considering this concept has little to do with what society consider as water resources (for domestic, industrial or environmental uses), virtual water is considered unhelpful and misleading to the discussion of water use in the livestock sector.

Land resource use was assessed using a number of land categories including cultivated arable land, non-cultivated arable land and non-arable land occupation. Data have not been reported by other researchers using these categories. Cultivated land occupation ranged from 0.9-9.3 m^2 / kg LW for the standard mid-weight market categories. Total land occupation (the combination of the three categories, was higher than most values reported in the literature for European beef

production which is not surprising considering the higher productivity in most regions of Europe compared to Australia. Land occupation measured in terms of 'total' land is of less relevance, particularly where results are used to compare productivity of ruminants grazing grass with monogastrics, which require grain grown on cultivated land.

Mid-weight export steers were standardised to 550 kg for grass finished scenarios or 620 kg for the grain finished scenario and represented the same age group of cattle. GHG emissions for the mid-fed steers ranged from 10.3-13.0 kg CO_2-e / kg LW. Emissions were higher from scenarios modelled under drought conditions. The emissions mitigation potential from grain feeding ranged from 9-21% for the mid-weight categories for the full life-time emissions of the animal. Compared to drought conditions, feedlot finishing resulted in a 28-30% reduction in GHG intensity. However, in general, there was a trade-off with higher energy use and cultivated land occupation to achieve this. In drought years, the trade-offs were much less apparent, because of the supplementary feed requirements. The project did not investigate the difference in land occupation impacts that may occur during drought conditions by removing cattle to feedlots compared to retaining them on grassland, though these impacts may be substantial.

One objective of this project was to re-evaluate LCA results for southern beef, reported in Peters et al. (2010a, b). The aim was to expand the scope of the work, improve the comprehensiveness of the analysis and use updated methods for impacts such as water use. The result of this re-analysis was an increase in the reported GHG emissions by 9-15%. Water use impacts were not fully comparable because of differences in the estimation method used. The more comprehensive assessment used in the current study (particularly the inclusion of evaporation from farm dams) generally led to higher water use estimates than previously, though they were still within the range reported by Peters et al. (2010b).

Recommendations are provided for future research, which could include investigation of a series of scenarios to reduce resource use and emissions intensity from grain fed beef.

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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

O_3 – Ozone

PFCs – Perfluorocarbons

SF_6 – Sulphur hexafluoride

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 has been 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.

The current 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. These are summarised for context as follows.

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 grass-fed beef operation;
- a southern NSW beef operation (grass-fed and feedlot finishing); and
- a Western Australian lamb operation.

Full details from this study can be found in the original reference (MLA report: Peters et al. 2009) or from the peer reviewed journal articles covering greenhouse gases / carbon footprint (Peters et al. 2010a), water use (Peters et al. 2010b) and nutrient management (Peters et al. 2011).

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 references: (Davis et al. 2008a, b).

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

This project focussed on the feedlot sector of the supply chain, investigating water, energy and greenhouse gas (total GHG) 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 objectives of the project were to:

- Quantify the environmental impacts from one northern NSW beef supply chain with cattle finished on grass or grain for three markets.
- To re-assess the environmental impacts of two southern supply chains previously used in COMP.094 with cattle finished on grass or grain for three markets.
- Provide credible data on water usage, energy usage, GHG emissions, carbon sequestration, land occupation, soil health parameters (including erosion) and eutrophication for the three supply chains.
- Incorporate indicators related to the production and consumption of human edible food and the broad role of the beef industry in providing high quality food products for Australian and overseas markets.
- Compare grain and pasture finishing in each supply chain.

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.

LCA differs from other environmental tools (e.g. risk assessment, environmental performance evaluation, environmental auditing and environmental impact assessment) in a number of significant ways. In LCA, the environmental impact of a product, or the function a product is designed to perform, is assessed. The data obtained are independent of any ideology and it is much more complex than other environmental tools. As a system analysis, it surpasses the purely local effects of a decision and indicates the overall effects (Peters et al. 2009).

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 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.

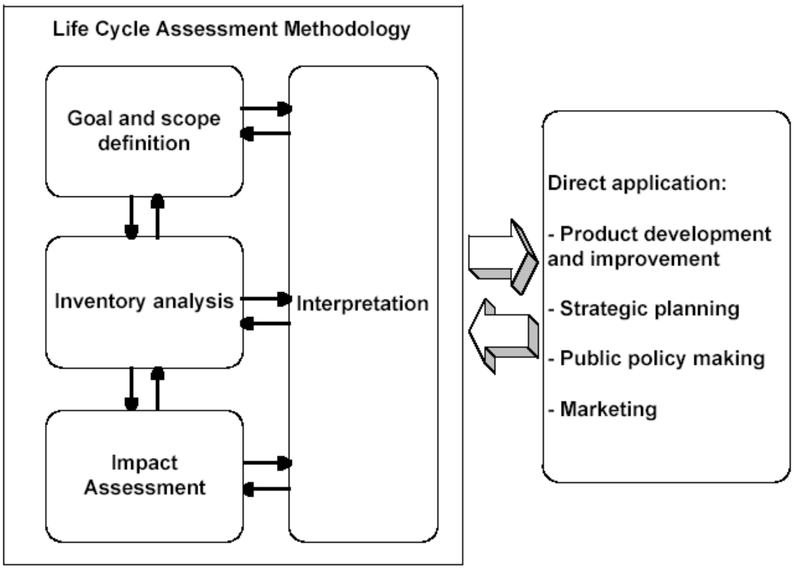

FIGURE 1 – GENERAL FRAMEWORK FOR LCA AND ITS APPLICATION (ISO 2006A: 14040)

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 of the strengths of LCA is it can develop broad answers long before the detailed research is completed. The broad scope of LCA (i.e. all greenhouse gases associated with a production system), which allows impacts to be 'classified' in terms of their overall impact, is also a strength. 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 protein 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’.

2.1.1 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.

2.1.2 Important methodological aspects of LCA research

2.1.2.1 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 should also be discarded to the environment without subsequent human transformation (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.

2.1.2.2 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, soil management, analyses etc.).

The background system covers other elements of the supply chain where data were 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.

2.1.2.3 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 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
FLOT.238 (Davis & Watts 2006), 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 focussed 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 on 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 over stretched. 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 world-wide. 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.

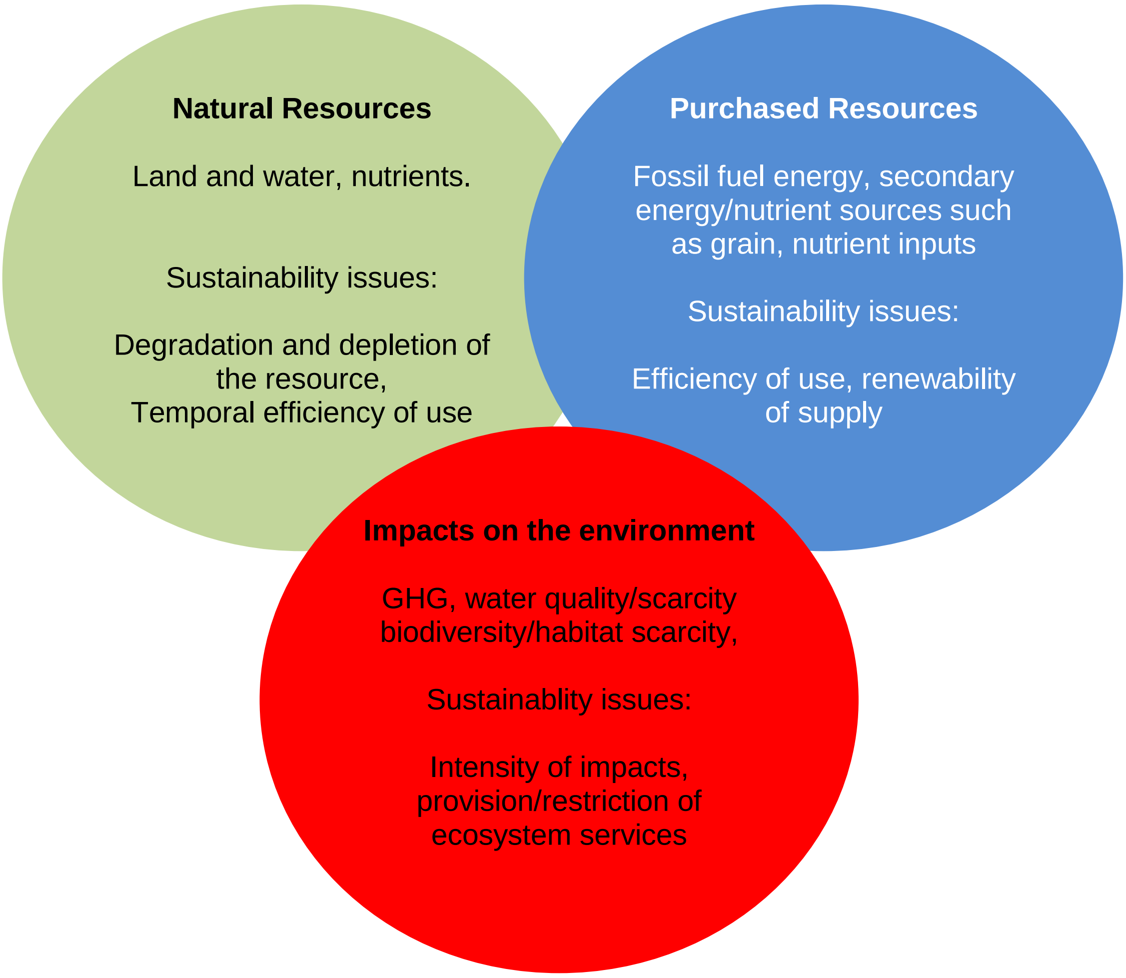

FIGURE 2 – KEY ELEMENTS OF AGRICULTURAL SUSTAINABILITY RELATED TO RESOURCE USE AND THE ENVIRONMENT

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 occupation, 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, taking natural restraints 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).

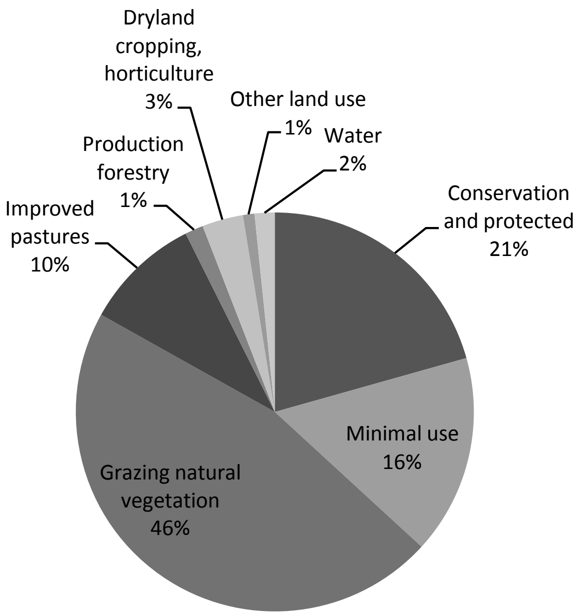

FIGURE 3 – MAJOR LAND OCCUPATIONS IN AUSTRALIA BASED ON THE 2005-06 DATASET (BRS 2010)

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 contrast, only 3.4% (0.26 M ha) of land is used for cropping. Hence, in Australia cropping land (arable land) is 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 occupation to another, though this is constrained by land type (soil, slope etc.), vegetation, annual rainfall, rainfall variability and evaporation. Land occupation 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 there was an objective to increase the protected area from 8.3 M ha to 20 M ha by 2020.

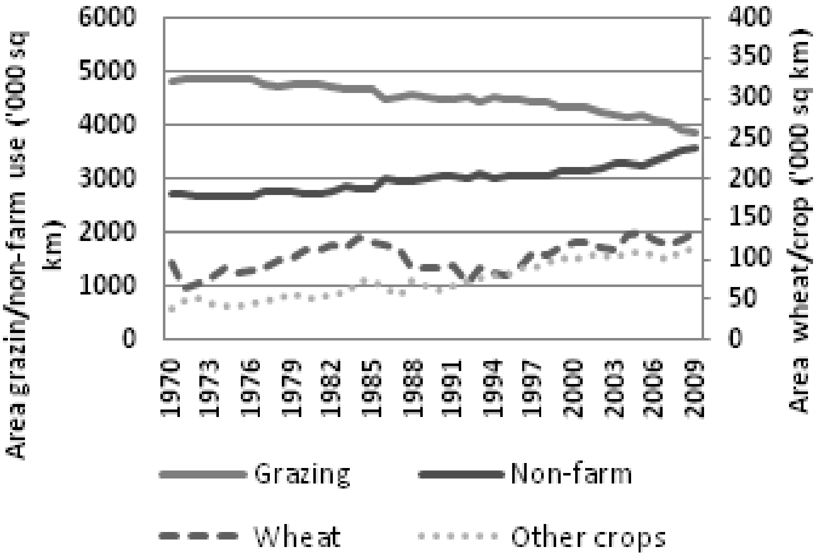

FIGURE 4 – TRENDS IN LAND OCCUPATION FOR MAJOR AGRICULTURAL PRODUCTION IN AUSTRALIA (ABARE 2009)

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 savannas 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 frame, measured in square meters (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. They also note that poultry and pigs require grain which could be fed directly to humans, while red meat production may not. Despite noting this, the study still compared "total land occupation", comparing livestock which may graze on non-arable land with poultry and pigs, which require grain grown from arable land. This should be seen as a major limitation to the usefulness of the findings.

Methodology development in the area of land occupation notes 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^2.yr) 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 OCCUPATION (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 Beef sourced from purpose grown beef herds	12.7 21.2
Pelletier et al. 2010	USA	Calves backgrounded on wheat pastures and finished in feedlot Calves finished on managed pasture and hay	84 120
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 to 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.

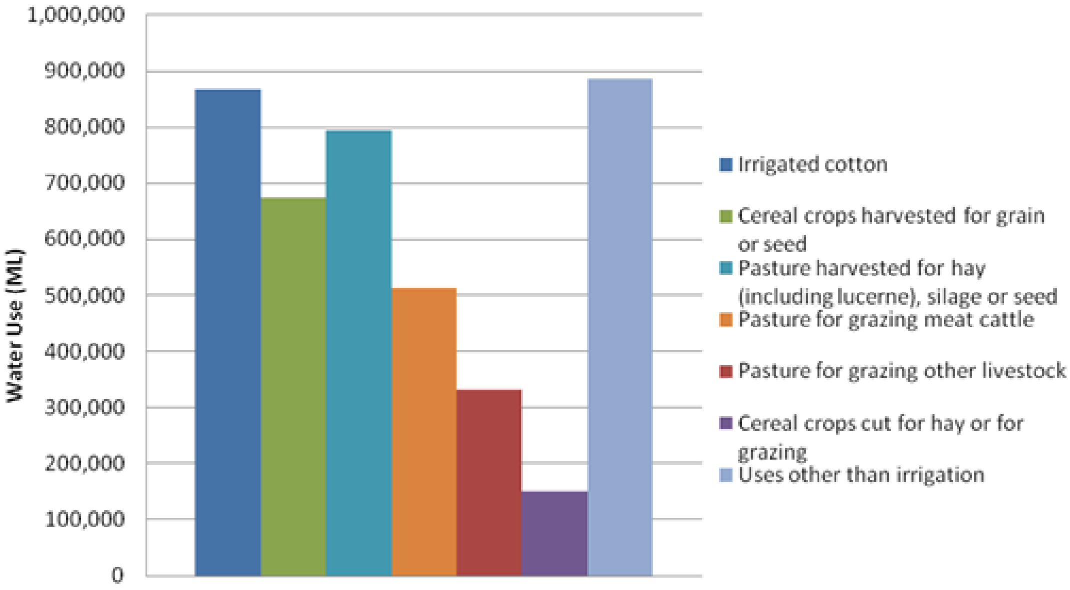

FIGURE 5 – WATER REQUIREMENTS FOR A NUMBER OF AGRICULTURAL COMMODITIES (ABS 2008)

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 issue 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 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 occupations (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 evapo-transpiration. In this situation, consumptive water use should be considered as the difference between runoff in the reference situation and the occupied land occupation. 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 liquid water that may be sourced 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, in 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 use

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 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. However, the efficiency of conversion for cattle 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.

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. McCaskill et al. (2003) identified no relationship between P fertiliser applications and P in surface runoff at the field scale for northern NSW, though a positive relationship was observed at four southern Australian sites.

The quantitative assessment was done using a risk assessment tool developed for Australian farms (the Farm Nutrient Loss Index, or FNLI – Melland et al. 2007). On each farm, two (or more) representative paddock types were selected and assessed against a set of standard criteria. From this site based assessment, risk ratings were determined for nutrient losses from each farm as a weighted average of the paddock scores.

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 visa-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 occupation (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 contribution (67.8% of national 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 may also arise from land use change (not assessed here).

Because of the dominance of enteric methane across the life cycle GHG emissions for beef

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

3.3.3.1 Enteric methane processes

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 were 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.

Beneficial findings have also been identified however. 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 has 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 conversation 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 the environmental impacts from four southern beef supply chains where cattle could be produced either from grass-fed or grain-fed finishing systems. Three market categories were identified, with either grain or grass finished cattle for each. These are as follows:

Australian Domestic

- Grass / forage finishing for domestic markets.
- Grain finishing for domestic markets (70 DOF).

Mid-weight export steers

- EU grass / forage finished cattle, 500-600kg LW and less than 24 mths of age.
- Mid-fed grain finishing for export markets (100-140 DOF).

EU grass refers to beef that is intended to be sold into the EU market. Mid-fed grain finishing refers to cattle that are finished in feedlots for an intermediate amount of time (100-140 days) and typically sold into export markets.

Heavy export bullocks

- Grass / forage finished to 600+ kg.
- Grain finished for export markets (300+ DOF).

The heavy export bullocks are most commonly sold into the Japanese market.

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, to reduce the complexity when presenting and interpreting the results, the primary functional unit selected for this report is **"one kilogram of live weight at the farm gate immediately prior to slaughter"**. This functional unit is most useful for comparing production systems through to slaughter and avoids the complexity of handling co-products (hides, offal etc.) at the meat processing stage.

When comparing alternative growing-finishing systems, the functional unit used was **'one kilogram of live weight gain over the growing-finishing stage'**. This is a partial supply chain functional unit (sometimes termed a 'gate-to-gate' functional unit) and does not represent the whole supply chain, but only the stage of relevance.

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). All supply chains had self-replacing breeding herds, with all impacts associated with breeding replacement bulls and heifers accounted for. Major components of the system are shown in Figure 6. This does not imply that other components were excluded.

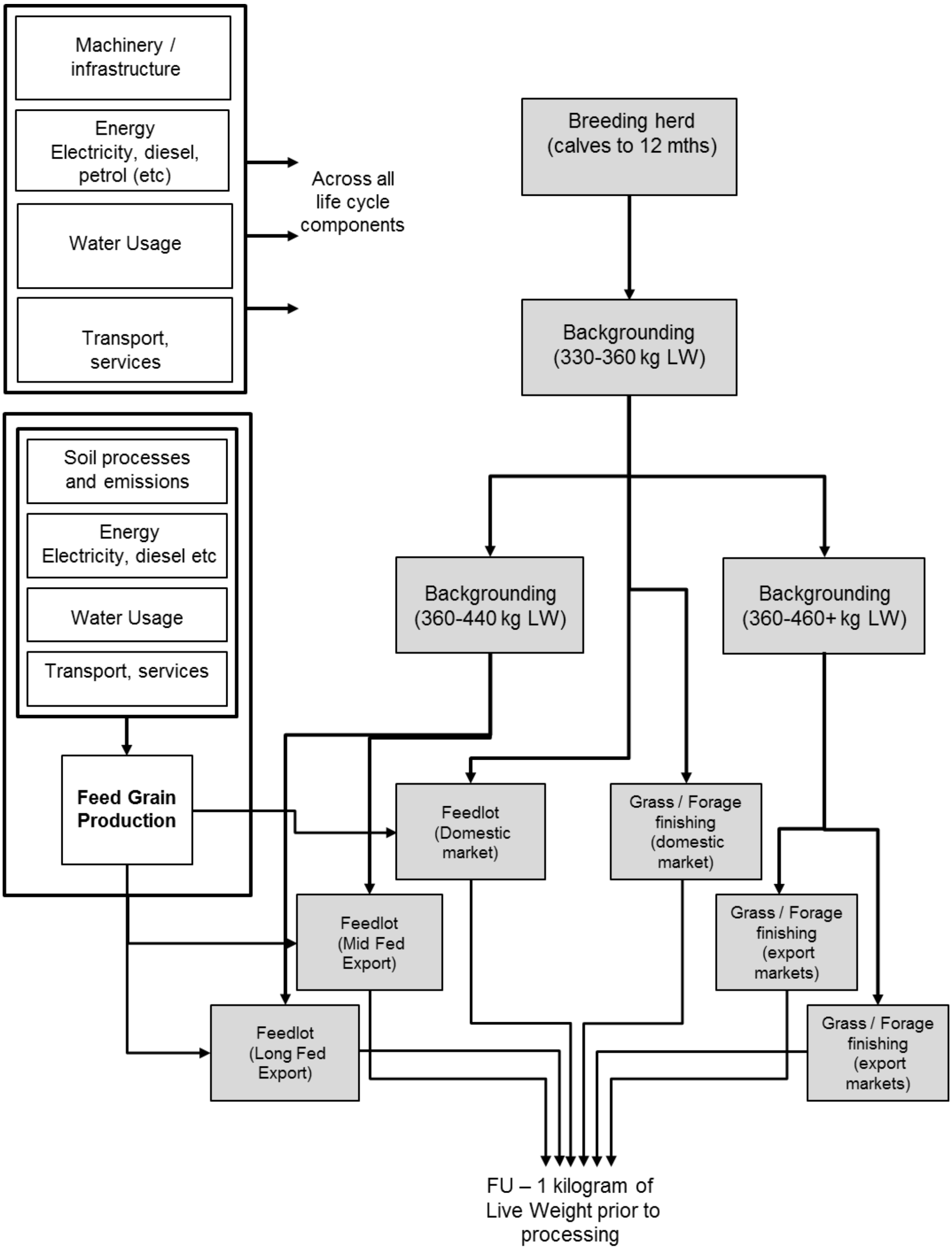

FIGURE 6 – SYSTEM BOUNDARY FOR GRASS AND FEEDLOT FINISHING SUPPLY CHAINS

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 based on estimation of Consumptive Fresh Water Use. We applied the impact assessment method 'stress weighted water use' (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 (occupation, measured in m^2 yr.) with three land occupation classifications i) use of non-arable pasture land, ii) use of arable land for pasture, and iii) use of cultivated arable land. A detailed explanation of the inventory methods and data are provided in Appendix 2 – Land occupation and nutrients.

4.3.4 Grain use

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. These data will be used to inform post-processing data analysis. At the farm gate level (the 'end point' of the present study) results will be presented for grain consumption only.

4.3.5 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. As the causes of eutrophication are quite different to many other regions in the world it was not appropriate to apply global characterisation factors. 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).

4.3.6 GHG emissions

GHG emissions were determined from all sources relevant to beef production 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 Global Warming Potentials that reflect the latest IPCC GWPs (see Table 8).

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 a number of beef properties and feedlots. Across the farms and feedlots cattle were being sold into a number of different markets that were broadly grouped into the domestic market (grain and grass fed), mid weight export cattle (i.e. EU grass and grain fed) and heavy export bullocks (Japan Ox, grass and grain fed). The project used a case study approach, and results could not be considered representative of 'NSW beef'.

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

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).

For the two supply chains involved in previous MLA projects, the data collection phase was not duplicated. However, all modelling processes were fully revised.

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 Case study farms

4.5.1 Northern NSW (Farm 1)

The first farm was located in a summer dominant rainfall zone, with annual average rainfall of 770mm and 65% of this falling in the six hottest months. Soils on the property were duplex (sandy loam over clay) granites (Grey Kurosols). These soils are naturally infertile, though the property had received phosphate fertiliser for over 40 years together with the introduction of legumes (white clover) and improved grasses (fescue and kikuyu) on approximately 70% of the farm, with the remaining area being native pastures and Eucalypt forest. Approximately 110 British breed cows are mated annually in a self-replacing herd. Surplus heifers are sold as yearlings for grass or grain finishing (domestic short fed) while steers are typically grass finished at 25-27 months and 530-540 kg LW. Steers may be sold to the domestic grass fed market or as feeder steers in some years. Characteristics of the breeding herd are provided in Table 9. The weaning rate for Farm 1 was higher than the five year (2006-2010) regional average of 85.4% reported for this region from 2006-2012 (ABARES 2013b).

TABLE 9 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE NTH NSW SUPPLY CHAIN (FARM 1)

Breeder cattle		
Production parameter	**Units**	**Average**
Weaning per cent	%	95.5
Breeder culling rate	%	13.0
Herd bulls	%	4.5
Mortality rate	%	1.5
Weaning weight (8 months)	kg LW	208.0
Backgrounding		
ADG (birth to 360 kg LW)	kg / d	0.65
Age at 360 kg LW	mths	16.7

4.5.2 Northern NSW (Farm 2)

Farm two was located in a summer dominant rainfall zone, with annual average rainfall of 660 mm. Soils on the property are predominantly low fertility brown clay loams (Brown Sodosols) commonly known as Traprock. The property was mostly open native pasture land, with sections of native forest. Vegetation on the property consisted of open eucalypt forest (common species being narrow leaf ironbark and spotted gum) and Cyprus. Grasses consist of native perennials (Queensland bluegrass, wiregrass grass) and naturalised species such as Coolati grass.

Approximately 210 cows are mated each year in a self-replacing herd. Calves are kept on farm to 10-11 months when they are transported to a second property for grass finishing or sold (either for grass or grain finishing). Replacement heifers are joined to calve at 2-3 years depending on the season.

The second property was located in a higher rainfall region (750 mm) on black earths (Vertisol) soils. Steers and heifers are grown out on native pastures with naturalised white clover, and finished on forage oats. Supplementary grain feeding is used in some years. Characteristics of the breeding herd are provided in Table 10. The weaning rate for Farm 2 was lower than the five year (2006-2010) regional average (85.4%) (ABARES 2013b).

TABLE 10 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE NTH NSW SUPPLY CHAIN (FARM 2)

Breeder cattle		
Production parameter	**Units**	**Average**
Weaning per cent	%	75.5
Breeder culling rate	%	20.2
Herd bulls	%	4.0
Mortality rate	%	1.5
Weaning weight (8 months)	kg LW	152.0
Backgrounding		
ADG (birth to 360 kg LW)	kg / d	0.61
Age at 360 kg	mths	17.8

4.5.3 Southern NSW (Farm 3)

The property was located in southern NSW, in a winter dominant rainfall zone with 550 mm annual rainfall. Soils on the property range from sandy loam granite soils to alluvial soils on river flats. The property was open grass land, with approximately 65% of the farm being sown with legumes (clover, lucerne) and receiving regular superphosphate fertiliser applications. A smaller portion (15%) of the farm is native pastures, while 20% is crop land used for growing grain and forage crops for livestock.

The farm was stocked with both sheep and cattle. Inputs specific to each enterprise were attributed accordingly. Where inputs could not be easily attributed to cattle or sheep, they were divided on the basis of the proportion of dry matter consumed by the sheep or cattle herds. On this basis, cattle were the major enterprise (73%).

Approximately 870 cows are mated each year in a self-replacing herd. Steers and sale heifers are sold to a number of markets including the domestic grass finished market (around 430 kg LW), mid-fed feedlot entry cattle (420-430 kg LW) or are grown out for heavy grass finished markets (520-530 kg LW or 600-610kg LW). Characteristics of the breeding herd are provided in Table 11. The weaning rate for Farm 3 was similar to the five year (2006-2010) regional average of 91.4% for this region (ABARES 2013a).

TABLE 11 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE STH NSW SUPPLY CHAIN (FARM 3)

Breeder cattle		
Production parameter	**Units**	**Average**
Weaning per cent	%	90.7
Breeder culling rate	%	15.0
Herd bulls	%	4.3
Mortality rate	%	1.6

Weaning weight (8 months)	kg LW	227.0
Backgrounding		
ADG (birth to 360 kg LW)	kg / d	0.82
Age at 360 kg LW	mths	13.2

4.5.4 South-Eastern Victoria (Farm 4)

The property was located in South-Eastern Victoria, in a winter dominant rainfall zone with 1030 mm annual rainfall. Soils are typically sandy loams (granite). The property was open grass land, with approximately 91% of the farm being pasture improved with white clover and ryegrass. A small portion of the farm (9%) was a fenced nature reserve.

The farm was a specialist beef producer, with approximately 300 cows being mated each year in a self-replacing herd. Sale heifers and steers were mainly sold to the domestic grass finished market (heifers – around 420 kg LW) and the heavy grass finished market (steers, 500 kg LW or 650-700 kg LW). Characteristics of the breeding herd are provided in Table 12. The weaning rate for Farm 4 was similar to the five year (2006-2010) regional average of 86.9% (ABARES 2013c).

TABLE 12 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THE EASTERN VIC SUPPLY CHAIN (FARM 4)

Breeder cattle		
Production parameter	**Units**	**Average**
Weaning per cent	%	85.7
Breeder culling rate	%	13.0
Herd bulls	%	2.8
Mortality rate	%	1.5
Weaning weight (8 months)	kg LW	239.0
Backgrounding		
ADG (birth to 360 kg LW)	kg / d	0.86
Age at 360 kg LW	mths	12.6

4.5.5 Feedlots – collected and modelled foreground data

Feedlot data were reported previously by Davis et al. (2008a, b). 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 13) 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. GHG emissions from manure management relied on these estimations. A detailed explanation of feedlot modelling methods can be found in the appendices.

TABLE 13 – PRODUCTION CHARACTERISTICS FOR THE THREE FEEDLOTS

Production parameter	**Units**	**Short Fed (Domestic)**	**Mid Fed (Export)**	**Long Fed (export)**
Entry weight	kg LW	360	421	442
Days on feed	days	63	115	335
ADG	kg / d	1.70	1.74	0.95
Exit weight	kg LW	467	622	761
Mortality rate	%	1.3	1.3	2.4

Daily Feed Intake	kg DMI	10.1	10.6	8.6
FCR		6.0	6.1	9.0

4.5.6 Standardised grass and grain finishing scenarios

In order to provide a comparison of grain and grass finishing, a series of scenarios were established based on data from the farms and feedlots. Most farms specialised in one or two different markets. However, where cattle were deemed to be appropriate for an alternative market to the one targeted by the farm, then these additional markets were also modelled. For example, farms selling EU grass finished cattle (Farm 1, Farm 4) were considered able to produce domestic grass finished steers at a slightly lighter weight (approx. 460 kg LW). All farms sold surplus heifers into the domestic grass finished market in some seasons, and sold heifers for grain finishing in other seasons. Cattle were predominantly British breed on all farms.

For each market (domestic, mid-weight export and heavy export) a minimum of three grass/forage scenarios were modelled along with the grain finishing scenario. The farms offered a reasonable spread of geography, land type and productivity. Livestock productivity data were reflective of average rainfall years (one below average year and one above average year). A further scenario was explored for the mid weight export cattle to represent a drought year (<75% of average annual rainfall). This was based on historical farm data for either the early or late 2000's. The drought scenarios were typified by lower growth rates and higher requirements for supplementary feeding. Livestock data are provided in Table 14 to Table 16.

TABLE 14 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THREE GRASS FINISHING SCENARIOS – DOMESTIC STEERS

Region	Finishing system	Growth rate to slaughter (kg/day)	Age at slaughter (mths)	Slaughter weight (kg)
Sth East Victoria (1000 mm a.a.r[a])	Clover / rye pasture, minimal external inputs.	0.76	18.7	460
Sth NSW (550 mm a.a.r)	Pasture backgrounding, supplementary grain feeding and irrigated forage finishing.	0.77	18.4	460
Nth NSW (770-800 mm a.a.r)	Pasture backgrounding and forage finishing.	0.76	18.5	460
Domestic Feedlot	Pasture backgrounding and feedlot finishing for 63 d	0.85	17.0	467

[a] average annual rainfall

TABLE 15 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THREE GRASS FINISHING SCENARIOS – EXPORT STEERS

Region	Finishing system	Growth rate (birth to slaughter (kg/d)	Age at slaughter (mths)	Slaughter weight (kg)
Sth East Victoria (1000 mm a.a.r[a])	Clover / rye pasture, minimal external inputs – average steer	0.77	22.2	550
Sth NSW (550 mm a.a.r)	Pasture backgrounding, supplementary grain feeding and irrigated forage finishing	0.59	28.8	550
Nth NSW (770-800 mm a.a.r)	Pasture backgrounding and dryland forage finishing.	0.66	26.1	550
Mid-fed Feedlot	Pasture and forage backgrounding, FL finishing for 115d	0.88	22.3	620
Drought Scenarios				
Sth NSW (550 mm a.a.r)	Pasture backgrounding and finishing, supplementary feeding – drought conditions.	0.48	35.7	550

Nth NSW (770-800 mm a.a.r)	Pasture backgrounding and finishing, supplementary feeding – drought conditions	0.48	35.7	550

[a] average annual rainfall

TABLE 16 – LIVESTOCK PRODUCTION CHARACTERISTICS FOR THREE GRASS FINISHING SCENARIOS – HEAVY EXPORT STEERS

Region	Finishing system	Growth rate (birth to slaughter (kg/d)	Age at slaughter (mths)	Slaughter weight (kg)
Sth East Victoria (1000 mm a.a.r[a])	Clover / rye pasture, minimal external inputs – heavy steer.	0.74	29.8	700
Sth NSW (550 mm a.a.r)	Pasture, supplementary feeding and forage finishing	0.54	34.6	600
Nth NSW (770-800 mm a.a.r)	Pasture backgrounding and forage finishing – heavy steer	0.64	34.3	700
Long-fed Feedlot	Pasture and forage backgrounding, feedlot finishing for 335d	0.79	30.6	761

[a] average annual rainfall

4.6 Handling co-production

Co-products were identified at two 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.

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 offcuts 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 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).

4.7 Impact assessment modelling

The LCIA 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 and discussion

The results section is divided into three sections. Results from the four breeding farms (cow-calf production) are presented '**per kilogram of live weight**' using a standardised 360 kg steer. Results for the average of the four farms are also presented, and these are used as the basis for the full supply chain analysis.

The second section compares the backgrounding and finishing phases for each market category. Results in this section are reported '**per kilogram of live weight gain**' during the backgrounding/finishing phase. These are partial supply chain results only.

The third section presents results for the full supply chains, comparing results from all market categories, based on the average steer process from the four farms. Results are presented '**per kilogram of live weight at the farm gate, immediately prior to slaughter**'. While referenced against the main output product (steers), the results represent aggregate beef production from steers and cull cows.

5.1 Cow-Calf production

5.1.1 Resource use

The resource use assessment covered energy, water, land and grain use. Most supply chains used relatively small amounts of energy (see Table 17) and there were no significant differences between the farms. The contribution to energy use was dominated by farm energy use (diesel, petrol and electricity use), purchased farm services (telecommunications, financial, repairs etc.), purchased fertiliser and pasture additive inputs, and use of supplementary feed.

Consumptive water use varied widely between the farms (see Table 17) as a consequence of different climatic factors driving drinking water consumption and differences in the water supply system. Drinking water use varied by > 2 fold between the farms, with the greatest differences seen between Farm 4 (49 L / kg LW) and farm 2 (116 L / kg LW). This was primarily as a result of climatic differences and to a lesser extent differences in herd efficiency. Consumptive water use (losses) from the water supply system varied to a much greater extent however, from -12 L / kg LW at Farm 4 to 226 L / kg LW at Farm 2. A number of factors governed consumptive water from the supply system. Firstly, each farm had a unique water supply profile. 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. Where water was supplied from dams (which contributed 20-85% of total water supply on the farms assessed) the consumptive use was heavily influenced by the water balance for the farm dams. The relative losses from farm dams was governed by two factors; i) the number of dams on each farm and the size of these dams relative to the number of livestock, and ii) factors effecting the water balance at each farm (rainfall, runoff and evaporation rate). Losses from the farm dams were determined by the volume of water intercepted from the environment compared to the reference system (i.e. the farm in the absence of dams). Therefore evaporative losses from dams were net of rainfall. This is explained further in Appendix 3 – Water use inventory.

The negative volume of water loss for the water supply at Farm 4 was unusual. This was because predicted evaporation was lower than annual rainfall, meaning that the dams gained water from rainfall within the banks of the dam compared to evaporation losses from the dam surface. Consequently, dams did not intercept water from the environment.
This was an unusual case however, and is restricted to regions with high rainfall and low evaporation rates. At Farm 4, this resulted in a lower consumptive water use value than the actual drinking water consumed by the cattle.

Farm 2 was located in a summer dominant rainfall zone with a relatively high degree of inter-annual variation. The farm had no permanent alternative water supply to farm dams and consequently the farm had a relatively high proportion of water storage per livestock unit compared to other farms, in order to manage the risk of water shortages in low rainfall years. This, combined with the high evapotranspiration rates for this site, resulted in very high evaporation rates compared to the other farms.

Stress weighted water use (see Table 17) was considerably lower for all supply chains with the exception of Farm 3, which was located in a water stressed region. Essentially this means water use from the Farm 3 supply chain has a much higher impact on the environment because of the relative scarcity of water in this region compared to the other farms.

TABLE 17 – ENERGY AND WATER USE FOR STEER AND HEIFER PRODUCTION FROM FOUR NSW BEEF CATTLE FARMS

Farm	Energy Demand	Consumptive Water Use	Stress weighted water use
	MJ / LW	L / kg LW	L H_2O-e / kg LW
Farm 1, Nth NSW	5.1 ± 12%	108.0 ± 21%	2.5 ± 29%
Farm 2, Nth NSW	5.0 ± 13%	342.0 ± 33%	12.4 ± 39%
Farm 3, Sth NSW	4.6 ± 13%	139.0 ± 10%	190.0 ± 16%
Farm 4, SE VIC	5.0 ± 13%	36.8 ± 19%	2.1 ± 18%
Average Cow/Calf Production	4.9 ± 13%	156.0 ± 21%	51.6 ± 21%

The land occupation assessment showed a distinct variation in cultivated land occupation (from 0.05 – 4.6 m^2 / kg LW), arable pasture land occupation (from zero to 51.6 m^2 / kg LW) and non-arable pasture land occupation (zero to 299 m^2 / kg LW). The difference in total land occupation (the combination of all categories) is driven by stocking rate, while the difference in land type reflects the resource base of the specific farms. Use of cultivated arable land was primarily driven by use of supplementary feed (grain and hay) which was highest on Farms 3 and 4. Arable land occupation for pasture was also highly variable (zero to 51.6 m^2 / kg LW). In most instances this land had been cultivated once or twice to sow pastures, but was never used for cropping.

Grain use was very low on most farms with the exception of Farm 3. This farm grew grain on farm for feeding over the summer. Grain use varied considerably from year to year across all farms in response to rainfall, grain prices and availability and management decisions (i.e. to sell cattle or supplementary feed).

TABLE 18 – LAND AND GRAIN USE FOR STEER AND HEIFER PRODUCTION FROM FOUR NSW BEEF CATTLE FARMS

Farm	Occupation of cultivated land (arable)	Occupation of pasture land (arable)	Occupation of pasture land (non arable)	Grain use
	m^2 / kg LW	m^2 / kg LW	m^2 / kg LW	kg / kg LW
Farm 1, Nth NSW	0.05 ± 12%	51.6 ± 15%	30.5 ± 17%	0.00 ± 4%
Farm 2, Nth NSW	0.43 ± 5%	0.0 ± 0%	299.0 ± 3%	0.01 ± 5%
Farm 3, Sth NSW	4.55 ± 0.2%	0.0 ± 0%	46.2 ± 13%	0.41 ± 41%
Farm 4, SE VIC	3.15 ± 6%	31.6 ± 16%	0.0 ± 0%	0.01 ± 6%
Average Cow/Calf Production	2.04 ± 22%	20.8 ± 11%	94.0 ± 3%	0.11 ± 39%

5.1.2 Environmental impacts

Impacts on land (Phosphorus Flux Potential, Soil Depletion Potential, Soil Carbon Flux Potential) and GHG emissions were assessed.

Phosphorus Flux Potential (depletion or accretion) was included as a measure of the long term sustainability of the system. Positive P flow values indicate a nutrient build-up while negative values indicate decline (Table 19). Of the four farms, two farms had declining levels of P (Farm 2 and 4) while two were positive (Farm 1 and 3). Positive P flux corresponded to the routine use of super phosphate on Farm 1 and 3. There were also small additions of P on all farms with supplementary feed. Ideally, grazing farms should maintain a long-term balance (i.e. close to zero) for phosphorus. Large additions of phosphorus will result in increased eutrophication risks, while depletion represents a decline in soil fertility which will result in lower pasture productivity and sustainability over time. Phosphorus is also a constrained global resource which needs to be managed into the future to ensure reserves are maintained. While negative values indicate that phosphorus is flowing out of the beef cattle system, this does not necessarily result in resource depletion. Where this phosphorus is retained in an accessible form and is utilised by other systems, this would be considered a transfer. However, if phosphorus contained in meat and by-products is lost to the ocean (via waste water treatment from cities) or is deposited in a land fill it could be considered resource depletion. This will be the subject of a later component of this research.

Soil erosion at two farms (Farms 2 and 3) contributed to the higher Soil Depletion Potentials compared to the other farms (Table 19). Soil depletion is a concerning environmental impact because of the severe consequences of excessive soil loss and the long timeframe required for soil formation.

TABLE 19 – PHOSPHORUS FLUX POTENTIAL AND SOIL DEPLETION POTENTIALS FOR STEER AND HEIFER PRODUCTION FROM FOUR NSW BEEF CATTLE FARMS

Farm	Phosphorus Flux Potential			Soil depletion Potential		
	kg P			kg		
Farm 1, Nth NSW	0.006	±	12%	0.00	±	59%
Farm 2, Nth NSW	-0.003	±	21%	63.40	±	41%
Farm 3, Sth NSW	0.006	±	56%	12.90	±	41%
Farm 4, SE VIC	-0.004	±	14%	0.31	±	60%
Average Cow/Calf Production	0.001	±	65%	19.20	±	39%

GHG emissions were driven by enteric methane (83-90%) and to a lesser extent by nitrous oxide emissions from manure (6-9%). Emissions from grain production and farm services were both minor, ranging from 0-4%, and 3-4% respectively. The contribution to GHG emissions is shown in Figure 7.

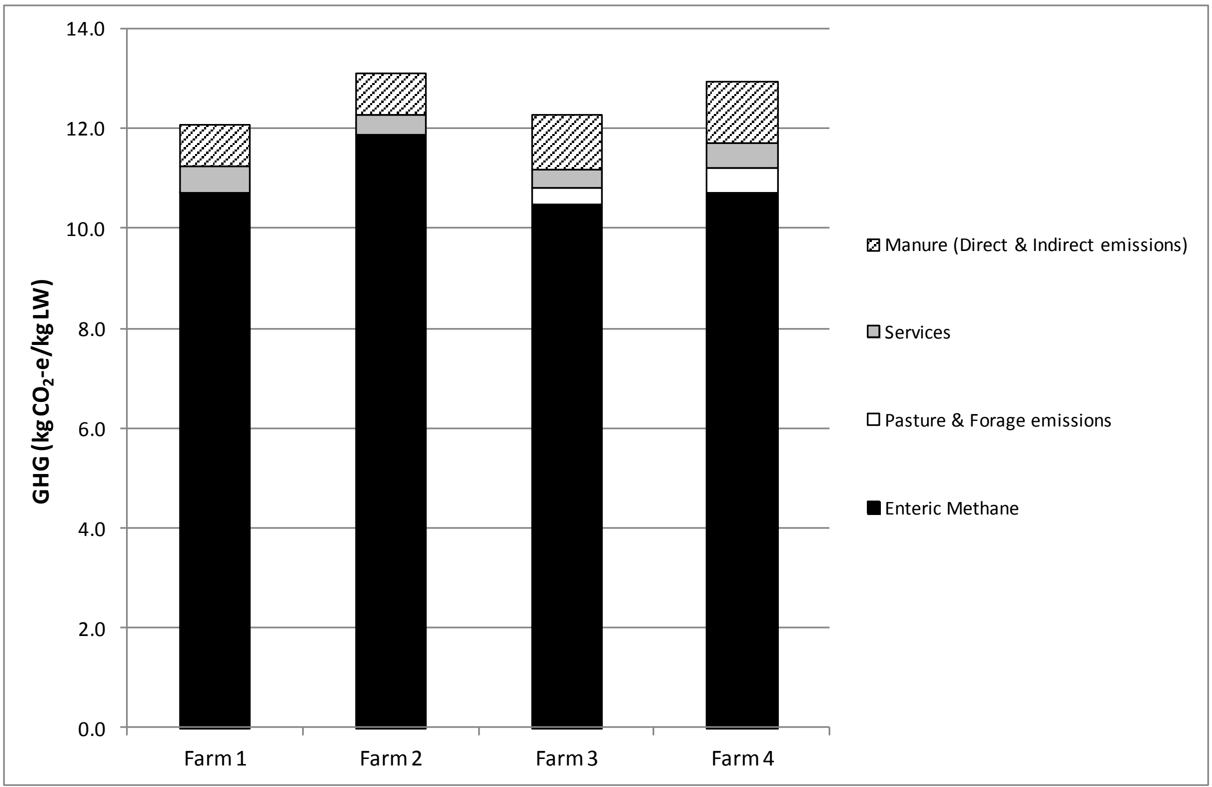

FIGURE 7 – GHG EMISSIONS FOR STEER AND HEIFER PRODUCTION FROM FOUR NSW BEEF CATTLE FARMS

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. 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 75-95 % across the four farms, while mortalities were relatively low on all farms (1-1.6%). Growth rates varied from 0.61 – 0.86 kg / d to 360 kg LW. The farm with the lowest herd productivity (Farm 2) generated the highest level of enteric methane emissions and overall emissions. However, this was partly compensated for by lower nitrous oxide emissions from pasture and manure (in response to the lower inclusion of legumes in the pasture and lower levels of leaching and runoff).

Soil Carbon Flux Potential is a measure of the change in soil carbon for soils throughout the supply chain. This ranged from negative (-1.9 kg CO_2-e / kg LW) to slightly positive (0.01 kg CO_2-e / kg LW). Negative values imply carbon sequestration while positive values indicate potential emissions of carbon from soils. This assessment was subject to a high degree of uncertainty (see Table 20) and sequestration potential from pastures is debated among scientists. The soil carbon sequestration rates under improved pasture were based on Chan et al. (2010). We assumed that soil carbon flux under native pastures was static, and that carbon losses (positive values) arose from cultivated land occupation, after Dalal & Chan (2001). The small positive flow from Farm 2 was associated with supplementary grain use.

TABLE 20 – SOIL CARBON FLUX POTENTIAL AND GHG EMISSIONS FOR STEER AND HEIFER PRODUCTION FROM FOUR NSW BEEF CATTLE FARMS

Farm	Soil Carbon Flux Potential	GHG emissions
	kg CO_2-e	kg CO_2-e
Farm 1, Nth NSW	-1.9 ± 78%	12.1 ± 18%
Farm 2, Nth NSW	0.01 ± 73%	13.1 ± 13%
Farm 3, Sth NSW	-1.1 ± 162%	12.3 ± 11%
Farm 4, SE VIC	-1.1 ± 143%	12.9 ± 19%
Average Cow/Calf Production	-1.0 ± 65%	12.6 ± 8%

5.2 Growing-Finishing systems

Results in this section are presented **"per kilogram of live weight gain (kg LWG) during the finishing process"** (i.e. excluding the impacts of the breeding herd). They do not represent the full impacts of beef production, which are presented in section 5.3.

The alternative backgrounding and finishing processes were developed from productivity and input data supplied by the farms. Each scenario was based on the average 360 kg steer from the four farms combined, with the exception of the export feedlots, which used a heavier steer (425-440 kg) averaged from the four farms. The average 360 kg steer was 14.9 months old, with a growth rate (birth to 360 kg) of 0.73 kg / d.

Market types reflected the data supplied from the original farms. Three farms sold export steers at 520-550 kg, and two farms sold heavy bullocks at ~700 kg. Typically, steers were sold at the maximum weight achievable at the end of the growing season (either the end of spring in the south, or late autumn in the north). This resulted in sub-optimal sale weights in many cases (i.e. 520-550 kg rather than 600-620 kg). Achieving heavier slaughter weights would require some form of intervention such as forage or grain feeding to improve growth rates over the lifetime of the animal, or keeping the animal for a substantially longer period of time.

5.2.1 Domestic steers

The four domestic market scenarios used forage finishing (Farms 2 and 3), pasture finishing (Farm 4) or grain finishing (domestic feedlot). Farm 3 also used supplementary feeding during the finishing period. The forage/grass finishing systems were standardised to a 460 kg LW (245 kg HSCW) animal, while the feedlot produced slightly heavier cattle (467 kg LW). The scenarios were modelled with steers, but surplus heifers could also be finished for this market. Heifers would be expected to have lower growth rates and lighter slaughter weights at a similar level of finish.

Resource use varied between the systems, reflecting the inputs required for production. Grass finishing (Farm 4) required the lowest inputs and had the lowest energy demand (2.9 MJ / kg LWG), while the forage finishing systems (Farm 2 and Farm 3) used slightly more energy (3.9-5.5 MJ / kg LWG respectively). Feedlot finishing required higher energy inputs (14.7 MJ / kg LWG), which was mainly 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 lowest (13.2 L / kg LWG) at Farm 4, because of the mild ambient temperatures, high rainfall and very low evaporation rates. Water use was considerably higher for Farm 2 (29 L / kg LWG) because of the hotter climate in Nth NSW. Farm 3 had a small irrigation licence which they used for irrigating forage, and this resulted in much higher water use (228 L / kg LWG), with 85% being contributed by the irrigation process. Water use was also quite high for the feedlot (113 L / kg LWG), with most of this (67%) being contributed by irrigated silage used in the ration.

Stress weighted water use was much lower than consumptive water use for Farms 2 and 4 (1.1-1.7 L H_2O-e / kg LWG) and for the feedlot (11.6 L H_2O-e / kg LWG). 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. Because each of these facilities was located in a region with relatively low water stress, the impact of using water was considered to be low compared to the global average water stress. Stress weighted water use at Farm 3 (313 L H_2O-e / kg LWG) in contrast, was higher than consumptive water use and considerably higher than any of the other farms. This reflected the higher level of water stress in the region where Farm 3 was located (around Wagga Wagga, NSW).

Cultivated land occupation showed a strong contrast between the grass finished, forage finished and grain finished scenarios. The grass finished scenario (Farm 4) used a very small amount of cultivated land (1.9 m^2 / kg LWG) associated with inputs such as hay and supplements, while Farm 2 (forage) used 23.8 m^2 / kg LWG and Farm 3 used 14.7 m^2 / kg LWG. In both cases this higher level of land occupation reflected the direct use of cultivated land for forage production. The trade-off between water use and land occupation was evident at farm 3, where finishing cattle on irrigation assisted forage resulted in much lower land occupation because of the higher yields in response to irrigation. Feedlot finished domestic beef used the highest amount of cultivated land (28.8 m^2 / kg LWG). This was similar to the dryland forage finishing scenario. Interestingly, the greater efficiency of grain finishing compared to forage finishing (FCR for the feedlot of 6, FCR for the forage crop of 9.3) only partly compensated for the lower average yields (t/ha) of grain production compared to forage.

Higher cultivated land occupation was associated with higher impacts from Soil Depletion Potential and soil carbon losses. It should be noted that these impacts have a high degree of uncertainty however. Phosphorus Flux was positive for two of the finishing systems (Farm 3 and the feedlot) reflecting the use of P fertilisers for forage and crop production. The grazing scenario (Farm 4) did not use P fertiliser however and had a P deficit (-0.005 kg / kg LWG), as did Farm 2, where P fertilisers are not regularly applied for forage production.

GHG emissions were lowest from the feedlot system (5.8 kg CO_2-e / kg LWG) and highest for the pasture finishing system (7.8 kg CO_2-e / kg LWG) – see Figure 8. The largest single contributor to GHG was enteric methane, which ranged from 43% for the feedlot to 84% for Farm 3. Contributions from manure emissions varied considerably, from 25% at the feedlot to only 6% at Farm 2 and Farm 3. The relatively high manure emissions at Farm 4 reflect the high levels of crude protein in the pasture diet, resulting in higher emissions. Contributions from feed production (ration commodities, forage or pasture) varied from 8-26% between the pasture and feedlot scenarios. At the feedlot, the use of grain to improve productivity is an effective strategy for reducing enteric methane, with the total impacts from grain production and transport being less than the comparative enteric methane emissions for grazing on grass. However, the higher CP levels in the diet for the feedlot compared to forage (16% for the feedlot ration, 11% for the forage crop) resulted in slightly higher manure emissions from the feedlot.

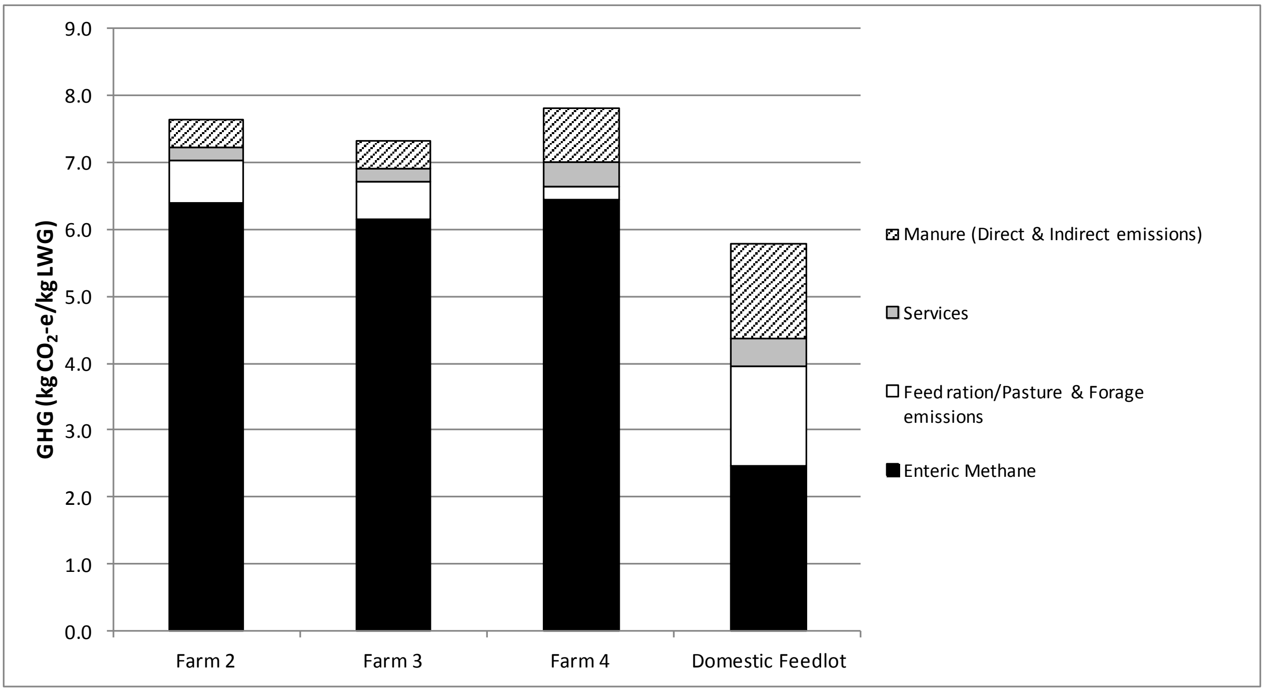

FIGURE 8 – GHG EMISSIONS PER KILOGRAM OF LIVE WEIGHT GAIN IN FOUR ALTERNATIVE FINISHING SCENARIOS FOR THE DOMESTIC BEEF MARKET

5.2.2 Mid-weight export steers

The mid weight export steers were standardised to 550 kg for the grass finished scenarios or 620 kg for the grain finished scenario. The 60 kg weight difference between grass and grain fed cattle afforded a slight efficiency improvement to the grain fed cattle by improving whole herd efficiency (kg beef / breeder cow). The feedlot steers represented the same age group of cattle, and the heavier weights are realistic considering the limitations of achieving higher weights from grass when managing seasonal feed shortages.

We also modelled two drought scenarios (for Farm 1 and Farm 3) in this weight category. Cattle in the drought scenarios had much slower growth rates and took 7-11 months longer to reach the 550 kg LW (see Table 15).

Energy use was low for the grass finished scenarios (4.3 MJ / kg LWG for Farm 1 and 3.1 MJ / kg LWG for Farm 4) where few inputs were utilised. These farms are both located in high rainfall zones, making low input pasture finishing realistic. Energy use was higher at Farm 3 (8.1 MJ / kg LWG) in response to the use of forage cropping and some supplementary feeding at this farm. Similarly to the domestic market, feedlot finishing reported the highest level of energy use (13.6 MJ / kg LWG).

Water use varied according to region in a similar fashion to the domestic market cattle. Consumptive water use estimates were very low on Farm 4 (14.1 L / kg LWG) and were higher on Farm 1 (77 L / kg LWG) mainly because of the higher dam evaporation rates. Water use was much higher at Farm 3 (642 L / kg LWG) where cattle were finished on irrigated forage crops. Water use at the feedlot was relatively low (47.2 L / kg LWG), reflecting the high level of productivity and absence of irrigation for feed production. Stress weighted water use was lower for Farms 1 and 4 (1.8 and 1.2 L H_2O-e / kg LWG respectively) because these farms were located in low water stress regions. Stress weighted water use was considerably higher for Farm 3 (873 L H_2O-e / kg LWG) because of the irrigation water use and the highly stressed region. Stress weighted water use was also higher than consumptive water use at the feedlot (57 L H_2O-e / kg LWG) because the feedlot was located in a water stressed region.

Cultivated land occupation was highly variable, reflecting the level of supplementary feeding across the farms. For the grass finished scenarios this was very low 0-2 m^2 / kg LWG (Farm 1 and Farm 4) while it was higher for Farm 3 (14.1 m^2 / kg LWG) where forage crops and supplementary feed (1.6 kg grain / kg LWG) were utilised. The highest rate of cultivated land occupation was from the feedlot (23 m^2 / kg LWG) as a result of grain feeding (4.1 kg grain / kg LWG). No other scenarios used any appreciable amount of grain. land occupation impacts such as soil depletion potential and soil carbon flux potential were highest for the farms that utilised cultivated land.

Across all impact categories, drought conditions led to poorer environmental outcomes. This was mostly as a result of poorer productivity, resulting in higher energy, water and land resources (with associated impacts) compared to the standard growth rate scenarios. Energy use was 64-76% higher for the drought scenarios, while water use was also 76% higher at Farm 1. We did not re-model dam water balances for the drought scenarios (these were based on long term averages), but higher evaporation to precipitation ratios during drought would result in much higher storage losses also. For Farm 3, we modelled a scenario where less water was available for irrigation in drought conditions, resulting in less water use compared to the standard scenario (530 v 642 L / kg LWG). Grain use was also considerably higher for the drought scenario at farm 3 (3.5 v 1.6 kg / kg LWG) because of the requirement to supplementary feed to assist growth rates and finishing. Interestingly, this level of grain feeding is approaching 4.1 kg grain / kg LWG used for lot feeding.

GHG emissions varied considerably between the scenarios. Where growth rates could be maintained at a high level on grass (such as on Farm 4) emissions were quite low. However, as growth rates declined, enteric methane emissions increased, reducing overall productivity (see Figure 9). The drought scenarios resulted in 35-75% higher emissions than the standard scenarios for Farms 1 and 3. Grain finishing resulted in a 29-57% reduction in GHG compared to the standard grass finishing scenarios, with emissions of 6.0 kg CO_2-e / kg LWG. Compared to drought conditions, feedlot finishing resulted in a 67-69% reduction in GHG emissions.

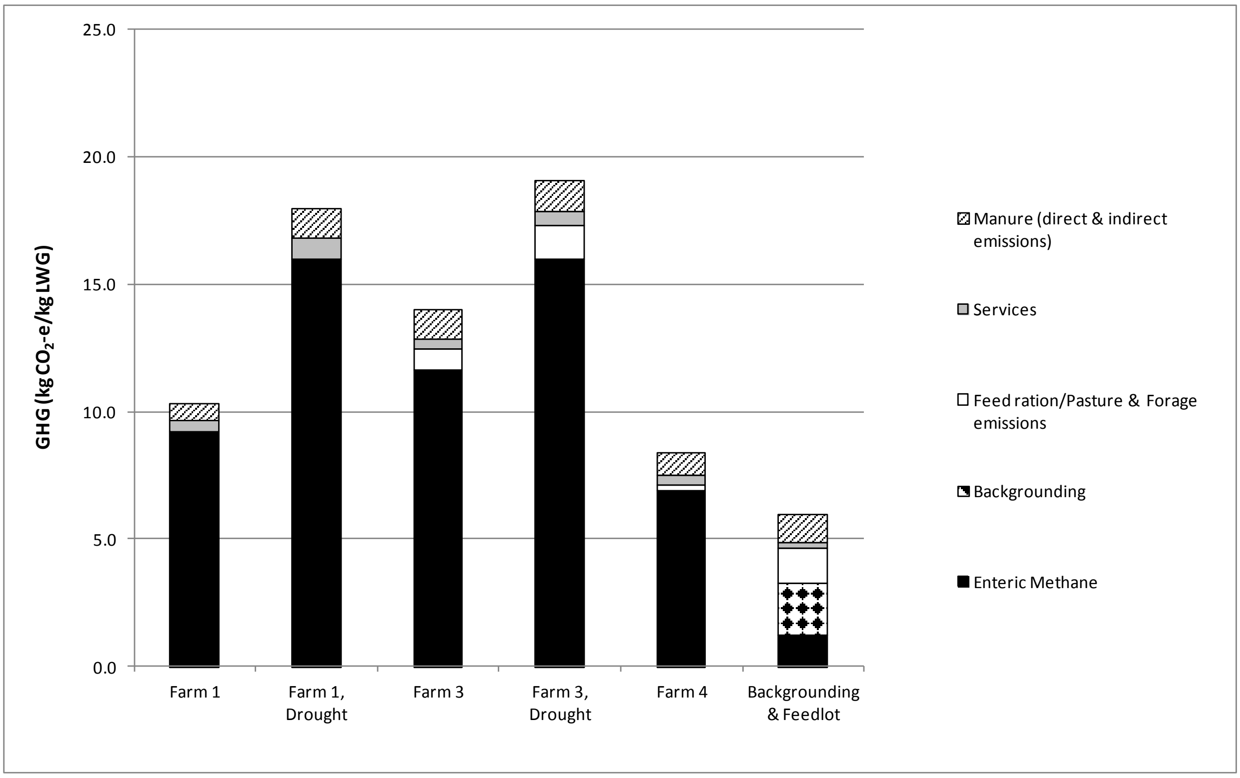

FIGURE 9 – GHG EMISSIONS PER KILOGRAM OF LIVE WEIGHT GAIN IN FOUR ALTERNATIVE FINISHING SCENARIOS AND TWO DROUGHT FINISHING SCENARIOS FOR MID-WEIGHT EXPORT STEERS

5.2.3 Heavy export bullocks

Four heavy export steer scenarios were modelled, one with grass backgrounding and finishing (Farm 4), and two with forage finishing (Farm 2 and Farm 3). These were compared with a long-fed feedlot scenario, where cattle were fed for 335 days for the premium Japanese grain-fed market. Production of heavy steers has benefits with respect to whole herd efficiency (maximising beef production / breeding cow) provided growth rates can be maintained at a fairly high level. As live weight increases, the maintenance requirements of the slaughter cattle also increase, making high growth rates more difficult to achieve.

Resource use reflected productivity and the production system. The high growth rate pasture/forage scenarios (Farms 2 and 4) had low levels of energy demand (4.2 and 3.9 MJ / kg LWG respectively) while Farm 3 had higher levels of energy demand (7 MJ / kg LWG) reflecting the greater reliance on grain and forage inputs. Water use varied with region following a similar trend to the domestic market class for these three farms (ranging from 17 – 102 L for Farm 4 and Farm 3 respectively). The long fed feedlot used higher levels of energy (17.3 MJ / kg LWG) and water (65 L / kg LWG). Stress weighted water use was very low for Farm 2 and 4 (2.6 and 1.5 L H_2O-e / kg LWG). As for the other market classes, stress weighted water use at Farm 3 (142 L H_2O-e / kg LWG) was higher than consumptive water use. Stress weighted water use at the feedlot (26 L H_2O-e / kg LWG) was mostly related to irrigation water used in the backgrounding process.

Cultivated land occupation varied considerably across the grass/forage scenarios, from 2.5 m^2 / kg LWG for Farm 4, to 11.6 m^2 / kg LWG for Farm 3. Cultivated land occupation was highest for the long fed feedlot (33 m^2 / kg LWG) which was driven by grain and hay/silage use.

Land occupation impacts were most heavily influenced by the use of cultivated land, which resulted in higher risks of soil depletion from erosion, and soil carbon losses. Full results are presented in the following section.

GHG emissions varied by 30% across the grass/forage scenarios, mainly in response to growth rate. Enteric methane emissions were highest from the grass/forage scenarios, while contributions from feed production and manure were higher from the feedlot scenario. Enteric emissions were highest from Farm 3 in response to lower growth rates (0.4 kg / d from 360-600 kg). Considering the lifetime growth rate for this scenario was 0.54 kg / d (see Table 16), these growth rates may be more representative of bullock production in southern Australia than the heavier, faster growing systems at Farm 2 and Farm 4. In comparison to Farm 3, emissions from the long-fed feedlot were 33% lower.

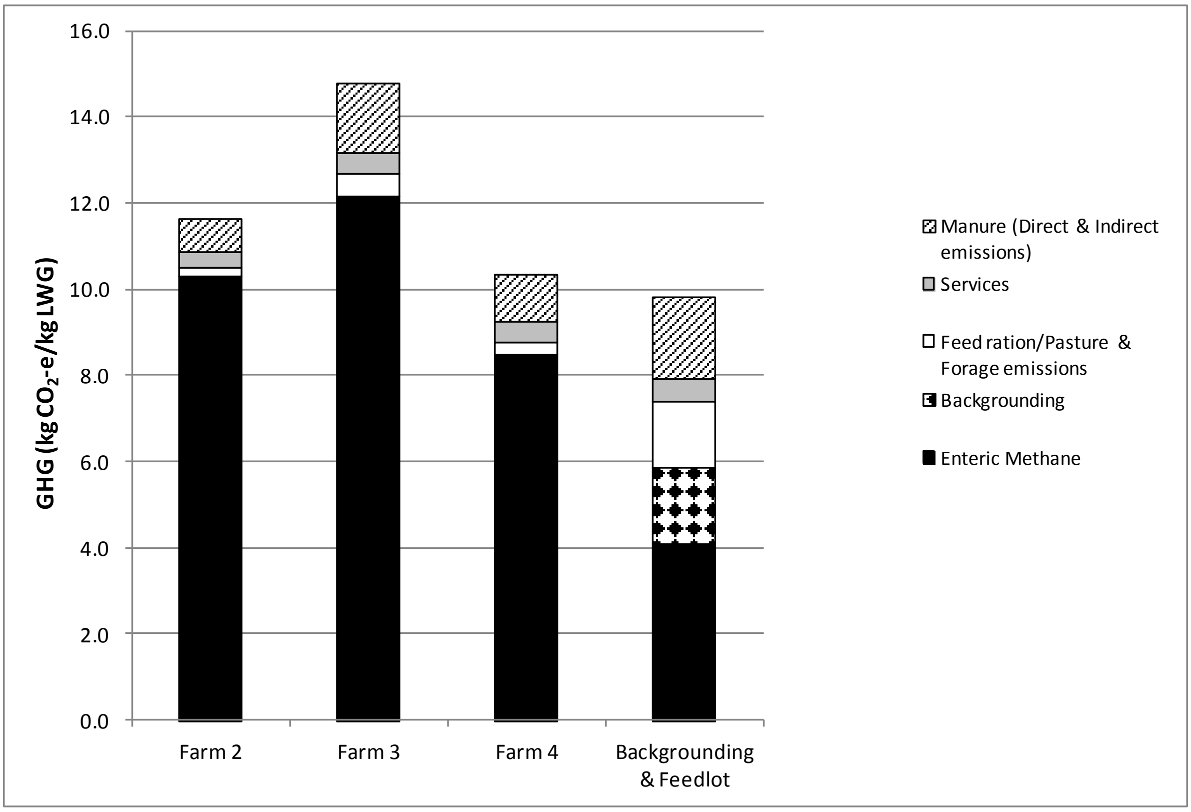

FIGURE 10 – GHG EMISSIONS PER KILOGRAM OF LIVE WEIGHT GAIN IN FOUR ALTERNATIVE FINISHING SCENARIOS FOR HEAVY EXPORT BULLOCKS

5.3 Analysis of alternative markets and finishing systems – full supply chain

To compare grain and grass finishing the full impacts across the supply chain must be taken into account rather than only the impacts during the finishing phase. Results in this section are presented **per kilogram of LW at the farm-gate**, for the full supply chain, including beef produced from the steers and cull breeding animals. The various finishing scenarios were described in the previous section where results were presented per "kilogram of gain". Results in this section include the breeding impacts to provide a complete picture and allow a 'full supply chain' comparison of grain and grass finishing.

5.3.1 Domestic steer

The domestic cattle are finished for the shortest amount of time, to lighter slaughter weights. Consequently, the finishing phase had a small impact on overall results, and differences between the finishing systems used were less apparent. The domestic market finishing phase was 63-116 d across the scenarios (12-20% of the animal's life span) and total live weight gain in this period was 22-23% of final slaughter weight.

Energy demand was not significantly different between the grass/forage finishing scenarios, though there appeared to be a slight trend towards higher energy use in the forage finishing systems. Feedlot finishing resulted in 47% higher energy use compared to the average of the grass/forage finishing scenarios (see Table 21). Consumptive water use ranged from 120.9 L / kg LW (Farm 4) to 160.5 L / kg LW for Farm 3. While the confidence intervals overlapped between the scenarios, a comparative Monte Carlo analysis comparing results (i.e. removing the influence of 'shared' variability) showed a significant difference between results in the majority of runs. The main difference between these systems was the use of a small amount of irrigation for the Farm 3 scenario and the low evaporation rates at Farm 4. The feedlot scenario used a relatively high amount of water compared to Farm 2 and 4 because this feedlot used some irrigation to produce silage. Stress weighted water use was lower than consumptive water use for all scenarios because the majority of breeding farms were located in low water stress regions. This said, stress weighted water use at Farm 3 was significantly higher than the other farms, because this farm used a proportion of irrigation water in a highly water stressed region. Interestingly, the relatively higher water use at the feedlot contributed little to stress weighted water use and results were therefore very similar to the lowest water users, Farm 2 and 4.

Cultivated land occupation varied from 1.3 m^2 to 6.5 m^2 between the grass finishing (Farm 4) and grain finishing scenarios. The forage finishing scenarios were intermediate to these values. Use of land from other categories (arable pasture, non-arable pasture) was reasonably static, as these land classes were used predominantly during the breeding and backgrounding phase. Grain use varied from very low (0.09 kg / kg LW) to 1.1 kg / kg LW for the feedlot. This represents the total grain required to produce beef from the whole supply chain. Considering most of the beef is produced from grass, the value for the feedlot scenario is still relatively low. The results were very sensitive to the use of grain feeding at any point in the supply chain. One farm (Farm 3) used grain to feed breeding cows over winter, and this contributed to the burden of grain used to produce the 'average' steer in all scenarios. Soil Depletion Potential was uniform across the alternative finishing scenarios. Phosphorus Flux Potential varied across the scenarios, being most strongly positive for the feedlot scenario. This reflects the use of grain inputs that are assumed to be grown with sufficient phosphorus fertiliser, compared to grazing and forage finishing scenarios where fertiliser was not routinely applied. Soil Carbon Flux Potential was negative (indicating carbon sequestration) across all the finishing scenarios, with the largest sequestration potential being on Farm 4 where cattle were finished on highly productive pastures. GHG emissions were not significantly different. Across all supply chains, enteric methane was the greatest contributor to GHG, followed by manure nitrous oxide emissions. Emissions from manure management and energy use were highest from the feedlot scenario (discussed in section 5.2.1).

TABLE 21 – RESOURCE USE FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR THE DOMESTIC MARKET

Farm	Energy Demand	Consumptive Water Use	Stress weighted water use	Occupation of cultivated land (arable)	Occupation of pasture land (arable)	Occupation of pasture land (non arable)	Grain use
	MJ / LW	L / kg LW	L H_2O-e / kg LW	m^2 / kg LW	m^2 / kg LW	m^2 / kg LW	kg / kg LW
Farm 2, Nth NSW	4.6 ± 9%	124.6 ± 17%	27.1 ± 22%	5.3 ± 2%	16.7 ± 12%	80.1 ± 3%	0.09 ± 23%
Farm 3, Sth NSW	4.9 ± 12%	160.5 ± 10%	82.1 ± 34%	3.6 ± 35%	16.6 ± 12%	83.1 ± 3%	0.27 ± 41%
Farm 4, SE VIC	4.4 ± 8%	120.9 ± 21%	26.8 ± 22%	1.3 ± 7%	19.9 ± 8%	79.6 ± 3%	0.06 ± 36%
SF Feedlot	6.8 ± 6%	139.6 ± 19%	28.7 ± 22%	6.5 ± 5%	16.5 ± 12%	79.2 ± 3%	1.11 ± 39%

TABLE 22 – ENVIRONMENTAL IMPACTS FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR THE DOMESTIC MARKET

Farm	Phosphorus Flux Potential	Soil depletion Potential	Soil Carbon Flux Potential	GHG emissions
	kg P	kg	kg CO_2-e	kg CO_2-e
Farm 2, Nth NSW	0.001 ± 65%	16.9 ± 40%	-0.7 ± 116%	12.4 ± 8%
Farm 3, Sth NSW	0.002 ± 53%	17.5 ± 40%	-0.8 ± 101%	11.7 ± 8%
Farm 4, SE VIC	0.001 ± 127%	16.4 ± 41%	-1.0 ± 69%	11.9 ± 8%
SF Feedlot	0.005 ± 28%	16.8 ± 39%	-0.7 ± 114%	12.1 ± 8%

5.3.2 Mid-weight export steers

The mid weight export steer scenarios showed more variation in resource use and impacts because of the relatively larger proportion of the lifespan and total live weight gain contributed by the backgrounding/finishing phase.

There was no significant difference in energy use between the standard grass and forage finishing scenarios. Energy use was significantly higher for the drought scenario at Farm 3 compared to the standard grass finishing scenarios, largely in response to the combined use of forage and supplementary grain feeding. Energy use was significantly higher for the grain finishing scenario compared to grass finishing, though these differences were not apparent when comparing the southern NSW drought scenario and the grain finishing scenario (see Table 23). Consumptive water use ranged from 107 L / kg LW (Farm 4) to 125 L / kg LW for Farm 1. The comparative Monte Carlo analysis comparing results (removing the influence of shared variability) showed a significant difference between results in the majority of runs. Water use was significantly higher at Farm 3 (298 L / kg LW) in response to irrigation water use, and these trends were similar for the stress weighted water use impact category, though the magnitude of water use was much lower for Farms 1 and 4. There was no significant difference in water use between the grass finishing and grain finishing scenarios.

Cultivated land occupation varied from 0.9 m^2 (Farm 1) to 5 m^2 (Farm 3) between the grass and forage finishing scenarios (see Table 23). Land occupation was significantly higher for the southern NSW drought scenario compared to the standard grass/forage scenario. Cultivated land occupation was higher for the feedlot scenario compared to the standard grass/forage finishing, but was lower than the southern NSW drought scenario.

Grain use varied from very low (0.1 kg / kg LW – Farm 1 and 4) to 1.5 kg / kg LW for the feedlot. Grain use was relatively low even for the feedlot scenario, as most of the beef is still produced in the grass finishing phases before grain finishing. Grain use was significantly higher for the Farm 3 drought scenario compared the standard grass/forage scenario in response to supplementary feeding in this system.

Phosphorus Flux Potential varied from negative to positive across the grass/forage scenarios depending on the use of phosphorus fertilisers. Phosphorus Flux Potential was most strongly positive for Farm 3. This reflects the use of grain inputs that are assumed to be grown with sufficient phosphorus fertiliser, compared to the other grazing and forage finishing scenarios where fertiliser was not routinely applied.

Soil Carbon Flux Potentials were negative (indicating carbon sequestration) across all the finishing scenarios, with the largest sequestration potential being on Farm 1 where stocking rates and pasture utilisation was lower than the southern farms, allowing the potential for more carbon return to soil.

GHG emissions varied across the finishing scenarios primarily as a result of differences in production efficiency (growth rate). The lowest values for grass finishing (Farm 4, Farm 1) were significantly lower than emissions from Farm 3 (see Table 24). There was also a significant difference between the standard scenarios and drought scenarios at Farm 1 and Farm 3. Emissions were lowest from the feedlot scenario (using comparative Monte Carlo uncertainty analysis) compared to Farms 1 and 4. Feedlot finishing showed significantly lower emissions compared to Farm 3 and when compared to the drought scenarios.

Across all supply chains, enteric methane was the greatest contributor to GHG, followed by manure nitrous oxide emissions.

5.3.3 Heavy export bullocks

Similar trends were evident in the comparison of the heavy export steers as the mid-weight steers for all impact categories. The high growth rates and high slaughter weights for the Farm 2 and Farm 4 grass/forage finishing scenarios showed a high level of resource use efficiency and low levels of impact intensity. These results were contrasted with Farm 3, where the slower growth rate and higher inputs resulted in higher GHG emissions (see Table 25).

Compared to the long-fed feedlot showed higher resource use and similar or lower GHG emissions from grain feeding (Table 26). The efficiency improvement for the feedlot compared to the grass finishing was not as apparent in this market class because the grass feeding scenarios had very high growth rates, while the performance in the feedlot is lower over this long feeding period compared to the domestic or mid-fed feedlots.

TABLE 23 – RESOURCE USE FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR MID WEIGHT EXPORT MARKETS

Farm	Energy Demand			Consumptive Water Use			Stress weighted water use			Occupation of cultivated land (arable)			Occupation of pasture land (arable)			Occupation of pasture land (non arable)			Grain use		
	MJ / kg LW			L / kg LW			L H_2O-e / kg LW			m^2 / kg LW			m^2 / kg LW			m^2 / kg LW			kg / kg LW		
Farm 1, Nth NSW	4.7	±	8%	125	±	20%	24	±	23%	0.9	±	10%	30.6	±	5%	69.0	±	3%	0.1	±	39%
Farm 1, Nth NSW, Drought	5.7	±	10%	144	±	16%	24	±	20%	0.9	±	10%	43.5	±	5%	69.5	±	3%	0.1	±	38%
Farm 3, Sth NSW	5.8	±	12%	298	±	41%	287	±	34%	5.0	±	35%	14.3	±	12%	79.2	±	3%	0.5	±	41%
Farm 3, Sth NSW, Drought	7.4	±	15%	266	±	35%	247	±	30%	10.2	±	37%	14.5	±	11%	83.5	±	3%	1.1	±	40%
Farm 4, SE VIC	4.3	±	8%	107	±	21%	23	±	22%	1.5	±	7%	20.2	±	8%	68.8	±	3%	0.1	±	36%
MF Feedlot	8.1	±	6%	111	±	19%	42	±	14%	9.3	±	6%	14.4	±	10%	63.9	±	3%	1.5	±	39%

TABLE 24 – ENVIRONMENTAL IMPACTS FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR MID WEIGHT EXPORT MARKETS

Farm	Phosphorus Flux Potential			Soil depletion Potential			Soil Carbon Flux Potential			GHG emissions		
	kg P			kg			kg CO_2-e			kg CO_2-e		
Farm 1, Nth NSW	0.002	±	34%	14.2	±	40%	-1.3	±	63%	11.9	±	8%
Farm 1, Nth NSW, Drought	0.005	±	25%	14.3	±	39%	-1.8	±	90%	14.3	±	8%
Farm 3, Sth NSW	0.003	±	53%	17.3	±	40%	-1.0	±	101%	13.0	±	8%
Farm 3, Sth NSW, Drought	0.005	±	42%	19.0	±	39%	-0.9	±	179%	14.8	±	8%
Farm 4, SE VIC	-0.0006	±	127%	14.2	±	41%	-0.9	±	69%	11.3	±	8%
MF Feedlot	0.002	±	9%	13.6	±	38%	-0.5	±	234%	10.3	±	7%

TABLE 25 – RESOURCE USE FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR HEAVY EXPORT MARKETS

Farm	Energy Demand	Consumptive Water Use	Stress weighted water use	Occupation of cultivated land (arable)	Occupation of pasture land (arable)	Occupation of pasture land (non arable)	Grain use
	MJ / kg LW	L / kg LW	L H_2O-e / kg LW	m^2 / kg LW	m^2 / kg LW	m^2 / kg LW	kg / kg LW
Farm 2, Nth NSW	4.6 ± 9%	123.0 ± 17%	19.9 ± 22%	4.1 ± 2%	11.7 ± 12%	73.9 ± 3%	0.07 ± 23%
Farm 3, Sth NSW	5.5 ± 12%	131.4 ± 41%	70.9 ± 34%	4.8 ± 35%	13.4 ± 12%	80.1 ± 3%	0.37 ± 41%
Farm 4, SE VIC	4.4 ± 8%	90.9 ± 21%	19.4 ± 22%	1.8 ± 7%	22.3 ± 8%	56.1 ± 3%	0.05 ± 36%
LF Feedlot	10.8 ± 6%	110.6 ± 14%	29.5 ± 13%	16.1 ± 6%	12.7 ± 10%	55.8 ± 3%	2.7 ± 39%

TABLE 26 – ENVIRONMENTAL IMPACTS FOR STEER PRODUCTION FROM FOUR ALTERNATIVE FINISHING SCENARIOS AND REGIONS FOR HEAVY EXPORT MARKETS

Farm	Phosphorus Flux Potential	Soil depletion Potential	Soil Carbon Flux Potential	GHG emissions
	kg P	kg	kg CO_2-e	kg CO_2-e
Farm 2, Nth NSW	-0.0006 ± 95%	12.0 ± 40%	-0.5 ± 116%	12.7 ± 8%
Farm 3, Sth NSW	0.0053 ± 53%	17.9 ± 40%	-1.1 ± 101%	13.4 ± 8%
Farm 4, SE VIC	-0.0008 ± 127%	11.6 ± 41%	-1.0 ± 69%	11.7 ± 8%
LF Feedlot	0.0140 ± 7%	12.6 ± 37%	-0.2 ± 388%	11.8 ± 7%

5.4 Eutrophication potential

Eutrophication could not be assessed because of the lack of regional characterisation factors for Australia, and the very different drivers of eutrophication compared to other regions. Findings from the qualitative assessment are informative none-the-less, and were done using a recognised nutrient loss risk tool. On all 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 (70-95%) and predominantly perennial pastures. Similarly, N and P loss via subsurface lateral flow was rated low for all farms. These scores are in good agreement with research for northern NSW (McCaskill et al. 2003) and southern NSW (Ridley et al. 2003, White et al. 2000) for similar stocking densities and fertiliser application rates.

Nutrient losses via deep drainage were rated as low for the first three farms and medium to high for farm 4. The low risk ratings relate to low annual rainfall and summer dominant rainfall (for the northern NSW farms), low nutrient application rates, the predominance of perennial pastures, and soil conditions. Farm 4, in contrast, was located a higher rainfall zone with winter rainfall, and experienced higher leaching rates. This, combined with the moderate levels of nutrients in the soil (from historical fertiliser applications) and nitrogen fixation from legume pastures, resulted in medium and high risk ratings for P and N accordingly.

TABLE 27 – EUTROPHICATION RATINGS FOR EACH FARM USING THE FARM NUTRIENT LOSS INDEX TOOL

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

The farm nutrient loss index was not suitable for assessing the nutrient loss risks from feedlots, because these farms have highly sophisticated systems for controlling the release of nutrients to the environment. All feedlots in the study maintained effluent containment systems and utilised nutrients at a rate suitable for forage or crop production. Uncontrolled releases were restricted to infrequent (<1 in 10 year) releases of low strength effluent when containment dams filled beyond their design capacity. Annualised nutrient losses from effluent releases to the environment were < 0.002 kg P / kg LW gain for the feedlot. These losses are at the point of discharge, which was (in most cases) several kilometres from the nearest stream. Consequently, feedlots were also considered to have a low eutrophication potential.

Aquatic eutrophication arising from diffuse nutrient sources on agricultural land is a concern in some, but not all river catchments in Australia. In river catchments were eutrophication is a concern, both phosphorus and nitrogen may be limiting nutrients and therefore both should be taken into account. Much of the eutrophication research in Australia has been carried out for high nutrient input dairy systems, or beef/sheep grazing systems in the south of the country (see Drewry et al. 2006). Dairy systems are much more likely to result in excess nutrient inputs and nutrient losses than beef grazing systems (Drewry et al. 2006). Similarly, grazing in southern Australia is more likely to result in nutrient losses than in the summer rainfall zones to the north of New South Wales, where two of the farms were located (McCaskill et al. 2003). While insufficient research was available to develop regionally specific characterisation factors for eutrophication within the scope of this study, the qualitative risk assessment process identified low levels of nutrient loss risk for the two farms located in northern NSW (farms 1 and 2) and the farm located in southern NSW (farm 3). All of these farms are located in the Murray Darling catchment, where the dominant nutrient source for eutrophication arises from gully and stream bank erosion, not from agricultural sources. Farm 4, while having the highest stocking rates and best livestock performance, also had the highest risk ratings for nutrient losses, particularly via the deep drainage pathway. This aligns well with research from this region, which has identified nutrient loss rates may be high for high input systems (Eckard et al. 2004). However, the very low levels of phosphorus and nitrogen inputs with fertiliser and additives to this property resulted in lower nutrient loss ratings that may have otherwise been the case.

While the method employed here to assess eutrophication potential does not allow direct comparison with the LCA literature, some contrasts are evident to other parts of the world. In contrast to Europe, stocking rates and fertiliser application rates are generally much lower in Australia (Davis & Koop 2006), resulting in less nutrients within the system, and nutrient balances that are neutral or negative for large parts of the country (NLWRA 2001). Eutrophication of inland river systems (relevant to most of the farms in this study) is primarily driven by gully and stream bank erosion, not by fertiliser additions to agriculture (Davis & Koop 2006), which is quite dissimilar to many other regions of the world where agricultural nutrients are strongly associated with eutrophication. Consequently, we consider the eutrophication potential from beef production to be very low for these farms. However, these findings may not be representative of cattle production in coastal catchments of Australia.

5.5 Comparison grass and grain finishing

The grass, forage and grain finishing scenarios reflect actual production conditions on four southern beef production farms and three feedlots. Two farms (Farm 2 and Farm 4) were located in high rainfall zones with good quality grazing land and productive pastures or forage crops. Consequently, cattle growth rates in these supply chains were high, particularly for the heavy bullocks. This may not be representative of many parts of NSW, and for this reason the results could not be seen as definitive.

The comparison of grain and grass finishing showed different results depending on the market class and depending on the impact category. Grain feeding resulted in improved herd efficiency, by maximising the total beef produced per tonne of dry matter intake across the whole herd. The principle drivers were improved growth rate (resulting in lower maintenance feed intake for slaughter cattle) and higher slaughter weights, which ‘dilutes’ the maintenance requirements of the whole breeding herd. This was most apparent in the mid-weight steers, which provided the maximum growth rate to slaughter, combined with heavy slaughter weights. The increased focus on greenhouse gas mitigation in Australia has led to greater attention on the productive efficiency of the national herd. While growth rate in slaughter cattle is known to reduce enteric methane emissions per kilogram of beef produced (Hunter & Niethe 2009), the ‘whole system’ impact of improving growth rates has not been previously assessed. Improving growth rates requires a higher plane of nutrition which is difficult to achieve at pasture for much of Australia. Grain feeding offers a more reliable means of improving growth rates. Our study investigated this, taking into account the impact of growing grain compared to growing pasture of forage, and the differences in manure management. The overall results showed distinctly lower GHG emissions for mid-weight cattle and negligible differences for domestic cattle or heavy export cattle for feedlot grain-fed compared to grass-fed beef. The results were sensitive to a number of assumptions. Firstly, the domestic grass/forage finished cattle were slaughtered at almost the same weight as the feedlot cattle, which may not be consistently achievable on grass or forage. Secondly, both major supermarkets are now taking grain fed cattle at heavier slaughter weights, and these cattle will be more efficient than the lighter grass fed cattle.

The most pronounced differences between grain and grass/forage finishing arose from the drought scenarios. In drought years, pasture quality and quantity decline, reducing growth rates and leading to higher requirements of grain or hay for supplementary feeding. This has a significant, adverse impact on GHG emission intensity. For the scenarios we modelled (which were based on data from the case study farms in drought years), we found drought feeding could result in much higher resource use (energy, cultivated land and grain) than standard grass/forage finishing, but these inputs did not provide the productivity improvement delivered by the feedlot because they were fed over a much longer period of time. GHG emissions were in the order of 67-69% lower for grain finishing (on a gate to gate basis) and 28-30% lower for the whole supply chain compared to supplementary feeding over an additional 7 months to achieve similar slaughter weights to the standard year. Decisions regarding drought feeding and management are difficult however, because it is never clear when the season will improve. With the benefit of hindsight, the most efficient (and quite possibly the most cost effective) strategy may be to sell cattle or lot-feed on farm in prolonged dry periods, but this is not clear at the time. Our analysis did not take into account the potential negative influences of grazing on land condition during droughts. The practice of removing these cattle from grazing lands and feeding through feedlots in drought years is widespread, and the benefits of this practice are likely to extend to impacts on land condition.

Trade-offs exist with feedlot finishing however. The higher productivity is achieved by using fossil fuel energy (for crop production, feed milling and transport) and the use of grain resources (and more fundamentally, cultivated land). While it may appear counter intuitive, it is common for agricultural systems to generate less GHG by utilising more energy, despite energy consumption also being a source of GHG emissions. This is because the savings in methane and nitrous oxide emissions outweigh the additional carbon dioxide emissions. The utilisation of additional resources enables herd efficiency to be maximised and emissions intensity to be minimised. The benefits to land quality have not been explored here but are likely to be significant. Few alternatives exist that do not result in similar trade-offs. Productivity could be consistently improved by irrigating pastures or forage crops, but this would also cause a trade-off between water use (much higher) and GHG (potentially lower).

In both the high rainfall grass finishing system (SE Victoria) and the feedlot finishing system there was a compensatory effect between enteric methane and manure nitrous oxide. In both systems, cattle were fed diets that had relatively high levels of crude protein well in excess of animal requirements. In the grass finishing operation, this was the result of the high estimated CP levels in legume pastures. The legume proportion in pasture is maintained partly to improve protein levels, but more importantly to boost pasture yield and digestibility. Higher digestibility pastures are essential for high growth rates, corresponding to lower methane intensity. Options may exist to limit this effect. Firstly, research is required to confirm several of the emissions pathways (particularly the indirect emission pathways from ammonia emissions, leaching and runoff, and the pasture residue emissions). If emission rates from these sources under Australian conditions are as high as suggested by the current, international default emission factors, it may be important to investigate the ideal proportion of legumes to maximise pasture yield and livestock production, with the lowest GHG emissions. Another potentially compensatory factor is the potential for soil carbon sequestration from these pastures. Sequestration requires high levels of pasture biomass production and deposition to the soil. This process is therefore linked to the processes generating nitrous oxide from these pastures. The current study did not use a detailed pasture modelling process, which would allow a better understanding of the impacts and interactions from these high performance pastures.

5.6 Comparison with Australian beef LCA results

Direct comparisons between LCA studies are difficult because of differences in system boundaries, handling of co-products methods and impact categories. However, results from other studies are useful for indicative purposes provided differences are taken into account. Three LCA studies have been conducted previously for Australian beef. Results for each of these studies are shown in Table 28.

TABLE 28 – GHG 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
Nth NSW, Sth NSW, SE VIC	Domestic Grass/forage finishing	11.7-12.4	121-161	This study
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	9.2-11 [a] (7.0 excl. breeding)	32 [a] (22 excl. breeding)	Peters et al. (2010a, b)
Sth NSW, inc. meat processing	Grain finishing (Mid-fed, 115d)	9.2-9.4 [a]	375-435 [a]	
Central NSW	Yearling (domestic grass)	10.4-10.6	24.7 - 167	CSIRO (Ridoutt et al. 2012, Ridoutt et al. 2011b)
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)	20.4 - 26.7	118-155	CSIRO Eady et al. (2011)
Arcadia valley, QLD	Jap Ox - grass-fed	13.7 - 18.2	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. Results have also been standardised using GWP values of 25 for methane and 298 for nitrous oxide.

The GHG results of the present study are in general agreement with previous studies, though GHG emissions tend to be slightly higher than Ridoutt et al. (2011) and Peters et al. (2010a). Ridoutt et al. 2012 and Ridoutt et al. 2011b, based their study on production and input data from NSW DPI gross margins rather than real farms. The case studies developed by Ridoutt et al. (2012; 2011b) had similar levels of herd productivity (weaning rates of 86% - four farms, 84% - one farm, and 64% - one farm) to our studies. Growth rates for grass finished slaughter cattle (calculated from slaughter weight and age, excluding birth weight) were 0.56-0.84 kg / d for Japan Ox (~ 640 kg LW), 0.59-0.68 kg / d for EU cattle (~530-570 kg LW) and 0.78-0.87 kg/d for the domestic cattle (~350-390 kg LW). These growth rates tended to be better than the growth rates observed for most of our farms, with the exception of Farm 4. Consequently, it is not surprising that the emissions intensity from these herds was slightly lower than ours.

Ridoutt et al. (2011) concluded that the benefits of lot feeding on reducing GHG could not be established from the study. However, the two feedlot scenarios modelled did not provide reasonable comparisons of grass and grain finishing because the breeding and backgrounding systems were not standardised. The feedlot finishing scenarios were based on herds with quite different levels of productivity (84% weaning, Walgett and 64% weaning, Casino) and had lower growth rates prior to cattle entering the feedlot (0.62 kg / d, Walgett-Gunnedah, 0.52 kg / d, Casino) than the grass finished scenarios modelled. Consequently, birth to slaughter daily gains for the feedlot finished cattle (0.76 kg/d) were less than some of grass finishing scenarios. The higher emissions for the cattle finished at Rangers Valley are therefore explainable from the poorer performance of the cattle herd and backgrounders prior to feedlot entry. Our consumptive water use results were similar to those reported by Ridoutt et al. (2012), though our results tended to be higher across all the supply chains considered. Excluding the impact of irrigation, and considering we applied the same drinking water estimation model as Ridoutt et al. (2012), the predicted evaporation rates from farm dams were much higher in our study. Ridoutt et al. (2012) did not collect data from actual farms, but applied a simple calculation to estimate the volume of water stored in farm dams. Ridoutt et al. (2012) also estimated stress weighted water use from the six farms, and showed a range of 3.3-221 L H_2O-e / kg LW. This is similar to the range in values we found in the present study for the breeding farms (2.1-190 L H_2O-e / kg LW). The average for the breeding farms (51.6 L H_2O-e / kg LW), which was used for all finishing systems, was considerably higher because of the influence of 'high stress' water use at farm 3. Consequently, stress weighted water use was higher across all of the finishing systems modelled (19.4-287 L H_2O-e / kg LW).

The results of our study also show higher GHG emissions than reported previously by Peters et al. (2010a). This is mainly because our study included a more detailed analysis across the supply chain. Our study included GHG emissions from grain production and manure emissions (including indirect emission sources) that were not accounted for previously. This study also used a GWP for methane of 25 as opposed to 24 which was ued by Peters et al. Perhaps the greatest difference was with the low values reported for the Victorian supply chain by Peters et al. (2010a), where impacts from the breeding herd were excluded. Considering the dominance of the contribution from the breeding herd, this value was not representative of the full supply chain impacts of beef production. Water use was assessed using a different metric in the current study compared to the previous. We also included farm water sources (i.e. drinking water from farm dams etc.) and the evaporation from these farm dams, which were not included previously. Despite these differences in approach, the overall results were relatively similar for one farm (sth NSW, Farm 3), mainly because we used the same data for irrigation, which was the largest contribution to water use on this farm.

The study by Eady et al. (2011) used high performing beef production systems from Queensland. The higher GHG emissions are mainly a result of differences in the enteric methane prediction method recommended by the DCCEE (2010) for Queensland compared to NSW. The DCCEE (2010) recommend an emissions estimation method based on Kurihara et al. (1999) for Queensland, which predicts much higher emissions than the method (Blaxter & Clapperton 1965) recommended for NSW applied in the present study. The very high values for the Gympie farm reflect the stage in the supply chain (weaner calves), which show naturally higher emissions because the enteric methane burden from the breeding herd is divided over less live weight. As demonstrated in the current study, emissions will decline as cattle grow to heavier weights provided growth rates are reasonable. It was not clear from the study what method Eady et al. (2011) applied for predicting evaporation from water storages, or if this was included at all. If evaporation losses were not included, the water use values are understandably lower.

5.7 Comparison with International beef LCA studies

A number of beef LCA studies have now been completed 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. We chose the mid-weight steer as the point of comparison. Where necessary, we converted results to a live weight basis using data from the studies and standardised the GWP values to 25 (methane) and 298 (nitrous oxide).

The results from our study are generally lower in energy use and similar to, or lower, in GHG emissions intensity than other studies in the literature. Energy demand for the mid weight steers in our study ranged from 4.3-8.1 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 this study reflects the low production intensity of Australian beef compared to Europe and the USA.

No studies were found in the international literature that reported water use using a sufficiently robust, full supply chain methodology. We did not attempt to compare with water footprint or virtual water use data as they use a very different method and have little relevance to the discussion of water use in the Australian beef industry.

Land occupation could only be compared as 'totals' which are of limited value. As expected, land occupation was higher in the present study 87-100 m^2 than most European studies, which ranged from ~22 m^2 / kg LW in the UK (Williams et al. 2006) to 26 m^2 / kg LW in France (Nguyen et al. 2012). However, our results were comparable to the estimates by Pelletier et al. (2010) for grain and grass fed beef production in the USA (84-120m^2 / kg LW)**.** 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. No studies were found that reported arable land occupation.

GHG emissions for the mid-fed steers ranged from 10.3-13.0 kg CO_2-e / kg LW.. European studies that investigated 'purpose grown' (i.e. non-dairy) beef production reported impacts in the order of 11-15 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 13.9 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.

6 Conclusions

This study is the first comprehensive LCA of southern beef production using data from case study farms. The objective of the study was to provide a comparison of finishing options (grass, forage and grain) using a standardised start and end point. To achieve this, the study modelled steer production from four grazing farms and averaged these processes. Because the breeding process is a substantial contribution to the whole supply chain, this approach reduced the variation that would have been seen had each farm been modelled independently. For each of the finishing processes, cattle were modelled from actual farm data for three different markets. The use of case study data means that the results could not be considered representative of 'southern beef production' 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 it did not include any herds with low productivity levels (the lowest weaning rate across the four farms was 75%). In the western districts it is likely that many herds would have lower weaning percentages than this, particularly in drought years. Similarly, growth rates for heavy grass fed cattle were quite high at two farms and could not be considered representative of all grass fed producers. With these limitations stated, a number of useful and important findings have come from this report.

Productivity Impacts on resources and environmental performance

This project demonstrated a general trend towards lower impacts from heavier slaughter cattle compared to lighter cattle, all factors being equal. This trend was driven by improved herd efficiency (kg beef / kg DMI across the whole herd) and was apparent while growth rates were high. Where growth rates declined, this trend was rapidly reversed as the maintenance requirements of the growing cattle increased with age. Achieving high growth rates may be difficult in grass finishing situations, and is even more challenging during drought years. One of the important roles the feedlot industry plays is improving supply chain productivity by maximising growth rates and slaughter weight. Trends towards higher slaughter weights in the domestic market should be welcomed as a move towards more efficient, more sustainable beef production.

The contrast between grass/forage and grain finishing was most apparent in the mid-weight export market modelling scenarios. Grain finishing significantly reduced emissions intensity compared to grass or forage finishing. Compared to the drought finishing scenarios, grain feeding resulted in a 28-30% reduction in GHG intensity. In general, there was a trade-off with higher energy use and cultivated land occupation to achieve this however. In drought years, the trade-offs were much less apparent, because significant amounts of grain, hay and forage (or failed cereal crops) were required to maintain and finish cattle. The study did not investigate the difference in land occupation impacts that may occur during drought conditions by removing cattle to feedlots compared to keeping them on grassland, though these impacts may be substantial.

Resource use and impacts

Energy use for the case study farms was low compared to other studies in the literature. The low input nature of the production systems, even when grain finishing was included, was the main driver of this. The results suggest water use is slightly higher on average than previously estimated by Australian studies, mainly because predicted evaporation loss from farm dams in the present study was much higher than estimated by other studies (if included at all). Few if any studies have comprehensively assessed water use using LCA internationally, and the difference in methodological approach precludes comparison with water footprint/virtual water results. Considering water footprint/virtual water data include rainfall used for pasture production, these type of studies provide little insight into water management in Australia. Our results are an order of magnitude lower than most published water footprint/virtual water use estimates because we have focussed on consumptive water use from water stored in rivers, dams, bores etc. and have excluded rainfall used to grow grass or crops fed to cattle.

The assessment of land occupation focussed on cultivated arable land resources, which are the most limited land resources in Australia and globally. No studies were found in the literature that differentiated between arable and non-arable land occupation, and this is considered a major flaw in the approaches taken to date, particularly where these results are used to compare with other livestock systems that are heavily reliant on grain and therefore cultivated land.

GHG emissions varied across the systems, primarily as a result of differences in productivity (discussed above) and differences in emission sources / rates from different systems. Enteric methane was by far the largest contributor to emissions across all supply chains. However, enteric emissions are much lower when cattle are fed grain, both in terms of daily emission rates (which may be 10-50% lower) and in terms of emissions intensity (CH_4 / kg LWG), with may be a factor 10 lower because of the higher growth rates on grain.

Noting the high degree of uncertainty, there appears to be an opportunity to offset emissions via soil sequestration in some instances. The likelihood of achieving soil carbon sequestration under pasture is subject to a high degree of debate among scientists. The results presented here are in no way conclusive, but do demonstrate the potential for emissions offset from this source, if conditions could be maintained that allow for sequestration. Conversely, the impact of carbon losses from cultivation was also taken into account and these two processes worked antagonistically (losses of soil carbon from cultivated land, gains on improved pasture land). Including soil carbon flux (generally losses) in the GHG assessment as part of LCA is increasingly expected, though most studies focus on the potential losses as a result of land transformation.

Comparison with previous Australian beef LCA research

One objective of this project was to re-evaluate LCA results for southern beef, reported in Peters et al. (2010a, b). The aim here was to expand the scope of the work, improve the comprehensiveness of the analysis, and use updated methods for impacts such as water use. The result of this re-analysis was an increase in the reported GHG emissions by 9-15% for the full supply chains. Water use impacts were not fully comparable because of differences in the estimation method used. The more comprehensive assessment used in the current study (particularly the inclusion of evaporation from farm dams) generally led to higher water use estimates than previously.

7 Recommendations

This study developed a significant knowledge base and modelling capacity, based on actual farm data. This provides some limitations and also opportunities for advancing knowledge in this area.

Having only four farms limits the representativeness of the study, and this could be rectified by expanding the case study dataset and applying a range of values based on industry performance. This could be achieved in a rapid and cost effective manner now that the modelling capacity and methodological issues have been addressed. The findings relating to differences between grain and grass finishing are potentially skewed by the relatively high growth rate grass finishing scenarios and this warrants further investigation.

Further research may be warranted to investigate the ideal production scenarios for maximising productivity and minimising resource use and environmental impacts from grain and grass finishing. Scenarios may include:

- Investigation of feeding cattle for a longer period on grain (either from weaning or from heavier entry weights)
- Investigation of a wider range of feeding periods and slaughter weights for lotfeeding and the impact on resource use and impacts.
- Investigation of GHG mitigation strategies at the feedlot, such as reduced dietary CP, improved FCR and energy / nutrient recovery from manure.
- Detailed investigation of options for finishing cattle in drought situations, including the full impact on land condition, compared to lotfeeding.
- Investigation of custom backgrounding operations with supplementary feeding to improve livestock growth rate prior to feedlot entry. This could assess impacts of cattle from weaning or yearlings through to slaughter.
- Investigation of growing out dairy bull calves or dairy cross calves as a source of low impact beef.

There are several outstanding impact areas that this project could be expanded to investigate. Importantly, the assessment of land occupation impacts was limited, and could be expanded to include other indicators of soil health such as acidification and salinity. This would expand the usefulness and comprehensiveness of the study. Soil acidification in particular would be relevant for many southern grazing enterprises.

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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 feedlots. 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.

In order to improve comparability between farms, the farm inventory data are presented here (Table 29 to Table 32) per five hundred kilograms of live weight sold off farm. This is approximately equivalent to a one sale animal and the total impacts associated with producing this animal from the whole herd and farm.

TABLE 29 – MATERIAL INPUTS AND OUTPUTS FOR FARM 1

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	8.5	1.06
Feed supplement		kg	1.6	1.06
Hay		kg	5.9	1.06
Drinking water	Data collected from farm	L	30 138.8	1.48
Land occupation	Data collected from farm			
Arable land (pasture, forage, grain)		m^2	26 352.2	1.20
Non arable (modified grazing pasture)		m^2	12 398.1	1.20
Energy	Data collected from farm			
Electricity		kWh	66.3	1.01
Oil		L	0.6	1.01
Diesel		L	7.0	1.01
Petrol		L	6.5	1.01
Transport	Estimated transport distances for cattle and farm commodities	t.km	184.9	
Fertilisers	Data collected from farm			
Superphosphate		kg	72.8	1.06
Other inputs and services				
Veterinary services		$	6.8	1.92
Communication services		$	5.5	1.92
Insurance		$	21.6	1.92
Automotive registration		$	10.8	1.92
Accounting		$	10.8	1.92
Banking		$	5.4	1.92
Industry levy		$	5.5	1.92
Outputs				
Cull cows		kg	70.3	
Excreted Manure				
Manure N	DCCEE (2010)	kg	134.0	
Emissions				
Enteric methane	DCCEE (2010)	kg	207.5	

TABLE 30 – MATERIAL INPUTS AND OUTPUTS FOR FARM 2

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	9.4	1.06
Feed supplement		kg	99.6	1.06
Hay		kg	65.0	1.06
Drinking water	Data collected from farm	L	56 410.9	1.45
Land occupation	Data collected from farm			
Non arable (modified grazing pasture)		m^2	135 723.9	1.20
Non arable (un-modified grazing pasture)		m^2	34 645.5	1.20
Energy	Data collected from farm			
Electricity		kWh	25.4	1.01
Oil		L	0.5	1.92
Diesel		L	10.3	1.01
Petrol		L	0.9	1.92
Transport	Estimated transport distances for cattle and farm commodities	t.km	115.4	
Other inputs and services				
Veterinary services		$	13.0	1.92
Communication services		$	5.8	1.92
Insurance		$	26.0	1.92
Automotive registration		$	11.3	1.92
Accounting		$	13.0	1.92
Banking		$	8.7	1.92
Industry levy		$	6.8	1.92
Outputs				
Cull cows		kg	182.4	
Excreted Manure				
Manure N	DCCEE (2010)	kg	142.7	
Emissions				
Enteric methane	DCCEE (2010)	kg	226.4	

TABLE 31 – MATERIAL INPUTS AND OUTPUTS FOR FARM 3

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	8.4	1.06
Hay		kg	15.7	1.06
Wheat		kg	210.2	1.06
Drinking water	Data collected from farm	L	32 909.2	1.48
Land occupation	Data collected from farm			
Arable land (pasture, forage, grain)		m^2	3 218.7	1.20
Non arable (modified grazing, pasture)		m^2	15 555.2	1.20
Non arable (un-modified grazing, pasture)		m^2	6 317.2	1.20
Energy	Data collected from farm			
Electricity		kWh	12.3	1.01
Oil		L	0.8	1.92
Diesel		L	7.7	1.01
Petrol		L	5.8	1.01
Transport	Estimated transport distances for cattle and farm commodities	t.km	21.7	
Fertilisers	Data collected from farm			
Single superphosphate		tonnes	0.1	1.01
Pesticides	Data collected from farm			
Lemat		g	14.0	1.04
Dimetheate		g	24.0	1.04
MCPA		g	209.0	1.04
Gramoxone		g	19.9	1.04
Other inputs and services				
Veterinary services		$	6.6	1.92
Communication services		$	5.4	1.92
Insurance		$	4.2	1.92
Automotive registration		$	2.1	1.92
Accounting		$	2.8	1.92
Banking		$	1.4	1.92
Industry levy		$	5.4	1.92
Outputs				
Cull cows		kg	95.9	
Excreted Manure				
Manure N	DCCEE (2010)	kg	198.6	
Emissions				
Enteric methane	DCCEE (2010)	kg	215.0	

TABLE 32 – MATERIAL INPUTS AND OUTPUTS FOR FARM 4

Inputs	Data source description	Units	per 500 kg LW	Uncertainty (SD or range)
Feed	Data collected from farm			
Pasture Dry Matter Intake		tonnes	7.9	1.06
Hay		kg	592.4	1.06
Molasses		kg	0.4	1.06
Drinking water	Data collected from farm	L	26 776.2	1.48
Land occupation	Data collected from farm			
Arable land (pasture, forage, grain)		m^2	15 282.7	1.20
Energy	Data collected from farm			
Electricity		kWh	6.4	1.01
Oil		L	0.8	1.92
Diesel		L	12.9	1.01
Petrol		L	5.8	1.01
Fertilisers	Data collected from farm			
Soil ameliorants		tonnes	1.4	1.06
Potassium Sulphate		tonnes	0.2	1.06
Transport	Estimated transport distances for cattle and farm commodities	t.km	307.7	
Other inputs and services				
Veterinary services		$	6.3	1.92
Communication services		$	5.1	1.92
Insurance		$	8.4	1.92
Automotive registration		$	4.2	1.92
Accounting		$	4.2	1.92
Banking		$	2.1	1.92
Industry levy		$	5.2	1.92
Outputs				
Cull cows		kg	82.7	
Excreted Manure				
Manure N	DCCEE (2010)	kg	207.2	
Emissions				
Enteric methane	DCCEE (2010)	kg	207.7	

Feedlot inventory data

Feedlot inventory data were collected over a three year period from detailed metering and monitoring of energy use, water use, commodity use and livestock numbers and performance. Full details are available from Davis et al. (2008a, b). 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. Financial records were confidential for all of the feedlots and were not available. These data were estimated from one feedlot where such data were provided and were allocated on a "per head day" basis across the feedlots.

TABLE 33 – MATERIAL INPUTS AND OUTPUTS FOR THE SF FEEDLOT

Inputs	Data source description	Units	per animal finished (360 - 467 kg LW)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	360.0	
Feed ration	Data collected from feedlot	kg DM	816.2	1.06
Land occupation	Data collected from feedlot			
Non arable (feedlot)		m^2	38.2	1.20
Arable (effluent irrigation)		m^2	208.2	1.20
Energy	Data collected from feedlot			
Electricity		kWh	5.7	1.01
LPG		L	0.0	1.01
Diesel		L	1.2	1.01
Petrol		L	0.7	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	72.0	
Other inputs and services				
	Veterinary services	$	9.4	1.92
	Communication services	$	0.2	1.92
	Accounting	$	6.5	1.92
	Industry levy	$	6.6	1.92
	Horse feed	kg	0.3	1.92
	Staff travel	km	0.6	1.92
Outputs				
Finished animal	Animal to abattoir	kg	467.0	
Excreted Manure				
Manure N	Mass Balance	kg	15.3	±10%
Manure VS	Mass Balance	kg	156.4	±10%
Manure P	Mass Balance	kg	2.2	±10%
Manure K	Mass Balance	kg	5.6	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979) and Beauchemin et al. (2008)	kg	12.6	

TABLE 34 – MATERIAL INPUTS AND OUTPUTS FOR THE MF FEEDLOT

Inputs	Data source description	Units	per animal finished (421 - 622 kg)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	421.0	
Feed ration	Data collected from feedlot	kg DM	1416.1	1.06
Land occupation	Data collected from feedlot			
Non arable (Feedlot)		m^2	12.6	1.20
Arable (effluent irrigation area)		m^2	12.9	1.20
Energy	Data collected from feedlot			
Electricity		kWh	3.2	1.01
Diesel		L	1.5	1.01
Petrol		L	1.2	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	84.4	
Other inputs and services				
	Veterinary services	$	17.0	1.92
	Communication services	$	0.3	1.92
	Accounting	$	11.9	1.92
	Industry levy	$	12.1	1.92
	Horse feed	kg	0.6	1.92
	Staff travel	km	1.0	1.92
Outputs				
Finished animal	Animal to abattoir	kg	622.0	
Excreted Manure				
Manure N	Mass Balance	kg	27.9	±10%
Manure VS	Mass Balance	kg	317.6	±10%
Manure P	Mass Balance	kg	3.5	±10%
Manure K	Mass Balance	kg	8.8	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979) and Beauchemin et al. (2008)	kg	21.4	

TABLE 35 – MATERIAL INPUTS AND OUTPUTS FOR THE LF FEEDLOT

Inputs	Data source description	Units	per animal finished (442 - 760 kg)	Uncertainty (SD or range)
Cattle	Data collected from feedlot	kg	442.0	
Feed ration	Data collected from feedlot	kg DM	3686.5	1.06
Land occupation	Data collected from feedlot			
Non arable (feedlot)		m^2	25.5	1.20
Arable (effluent irrigation area)		m^2	70.4	1.20
Energy	Data collected from feedlot			
Electricity		kWh	10.1	1.01
Diesel		L	11.3	1.01
Petrol		L	1.1	1.01
Transport	Estimated transport distances for cattle and feedlot commodities	t.km	88.7	
Other Purchases and inputs (expenses)				
	Veterinary services	$	49.6	1.92
	Communication services	$	0.9	1.92
	Accounting	$	34.5	1.92
	Industry levy	$	35.2	1.92
	Horse feed	kg	1.8	1.92
	Staff travel	km	3.0	1.92
Outputs				
Finished animal	Animal to abattoir	kg	760.8	
Excreted Manure				
Manure N	Mass Balance	kg	75.9	±10%
Manure VS	Mass Balance	kg	772.9	±10%
Manure P	Mass Balance	kg	11.3	±10%
Manure K	Mass Balance	kg	41.6	±10%
Emissions				
Enteric methane	Modelled from feed data using Moe & Tyrell (1979) and Beauchemin et al. (2008)	kg	67.0	

Feed milling and rations

Feed milling inventory data for all feedlots were based on records kept by the feedlot. These data are presented in Table 36.

TABLE 36 – MAJOR INPUTS FOR FEED MILLING AT THREE AUSTRALIAN FEEDLOTS

Inputs	Data source description	Units	SF feedlot (per tonne delivered to bunk)	MF feedlot (per tonne delivered to bunk)	LF feedlot (per tonne delivered to bunk)
Energy					
Electricity	Data collected from feedlot	kWh	7.2	7.1	6.4
LPG		L	n.a	6.0	n.a
Butane		m^3	n.a	n.a	1.7
Diesel		L	2.3	0.74	2.4
Water	Data collected from feedlot	L	48.3	205.1	96.8
Transport	Est. transport distances for commodities to the feedlot	t.km	194	247	175

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 three years. 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 37 to Table 39 show the aggregated, simplified rations for the three feedlots.

TABLE 37 – AGGREGATED, SIMPLIFIED RATIONS FOR THE SF FEEDLOT

Commodities (protein content in brackets)	Units	Amount
Maize (8%)	kg	91.8
Wheat (13%)	kg	513.1
White fluffy cottonseed	kg	102.3
Hominy Meal	kg	56.7
Lucerne Hay	kg	14.4
Corn Straw	kg	29.4
Maize Silage	kg	100.7
Molasses	kg	32.1
Supp Fin 6% Wet	kg	59.6
Total	kg	**1000.0**

TABLE 38 – AGGREGATED, SIMPLIFIED RATIONS FOR THE MF FEEDLOT

Commodities (protein content in brackets)	Units	Amount
Barley (10%)	kg	100.2
Wheat (13%)	kg	512.7
White fluffy cottonseed	kg	121.9
Lucerne Hay	kg	50.9
Wheat Hay	kg	3.7
Cotton Hulls	kg	72.4
Tallow	kg	37.7
Feed additives	kg	100.6
Total	**kg**	**1000.0**

TABLE 39 – AGGREGATED, SIMPLIFIED RATIONS FOR THE LF FEEDLOT

Commodities (protein content in brackets)	Units	Amount
Barley (10%)	kg	40.4
Maize (8%)	kg	35.0
Wheat (13%)	kg	410.0
Sunflower (36%)	kg	2.9
Cereal straw	kg	132.0
Maize Silage	kg	122.5
Pasture Silage	kg	23.2
Wheat Silage	kg	16.5
DDG (dried distillers grain)	kg	57.6
Mill Mix	kg	57.3
Molasses	kg	79.5
Feed additives	kg	23.6
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.

Appendix 2 – Land occupation and nutrients

Land occupation

land occupation was divided into three classifications; i) arable land (land occupied 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. These data were used to assist in modelling pasture N residue deposition rates.

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 2001a). 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 most sites erosion rates were considered to be at the lower end of the scale identified by the NLWRA mapping. At one farm (Farm 2) the NLWRA erosion data were not considered representative of current management. This was primarily because management at the farm had altered significantly (shift from high stocking rates with sheep to low-moderate stocking rates with cattle) since the NLWRA assessment in the late 1990s to early 2000s. Consequently, erosion rates were significantly reduced for this farm. Data are reported in Table 40 to Table 46.

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 (ie 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. Leaching losses were negligible at the northern sites, but was evident at the southern sites. Indirect nitrous oxide emissions which occur as a result of nitrogen loss in runoff were predicted using an emission factor of 0.0125 kg N_2O/kg N in runoff (DCCEE 2010). Table 40 to Table 46 show the parameters used to calculate sediment loss and nitrogen loss in runoff for each of the farms investigated in this study.

TABLE 40 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 1 (CULTIVATED LAND & PASTURE – ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	777	-	BOM (2012)
Runoff fraction	-	0.07	-	CSIRO (2007)
N in runoff and subsurface flow	kg/ha/yr	0	-	
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0	-	ArcGIS with NLWRA (2001b)

TABLE 41 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 1 MODIFIED GRAZING (PASTURE – NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	777	-	BOM (2012)
Runoff fraction	-	0.07	-	CSIRO (2007)
N in runoff and subsurface flow	kg/ha/yr	0	-	
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	0	-	ArcGIS with NLWRA (2001b)

TABLE 42 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 2 MODIFIED GRAZING (PASTURE – NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	661	-	BOM (2012)
Runoff fraction	-	0.05	-	CSIRO (2007)
N in runoff and subsurface flow	kg/ha/yr	0	-	
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	2.17	± 50%	ArcGIS with NLWRA (2001b)

TABLE 43 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 2 UN-MODIFIED GRAZING (PASTURE – NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	661	-	BOM (2012)
Runoff fraction	-	0.05	-	CSIRO (2007)
N in runoff and subsurface flow	kg/ha/yr	0	-	
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	2.17	± 50%	ArcGIS with NLWRA (2001b)

TABLE 44 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 3 MODIFIED GRAZING, (PASTURE – NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	559	-	BOM (2012)
Runoff fraction	-	0.06	-	CSIRO (2008b)
N in runoff and subsurface flow	kg/ha/yr	0.8	0.4-1.2	Ridley et al. (2003)
N lost through leaching	kg/ha/yr	6.0	0-19	White et al. (2000)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	2.70	± 50%	ArcGIS with NLWRA (2001b)

TABLE 45 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 3 UN-MODIFIED GRAZING, (PASTURE – NON-ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	559	-	BOM (2012)
Runoff fraction	-	0.06	-	CSIRO (2008b)
N in runoff and subsurface flow	kg/ha/yr	0.4	0.2-0.6	Ridley et al. (2003)
N lost through leaching	kg/ha/yr	3	0-9.5	White et al. (2000)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr	2.70	± 50%	ArcGIS with NLWRA (2001b)

TABLE 46 – PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS FOR FARM 4 (PASTURE – ARABLE)

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	936	-	BOM (2012)
Runoff fraction	-	0.20	-	CSIRO (2008a)
N in runoff and subsurface flow	kg/ha/yr.	3.9	1.95-5.85	Ridley et al. (2003)
N lost through leaching	kg/ha/yr.	14.6	7.3-21.9	Eckard et al. (2004)
Difference in soil erosion rate for pre-European and post-European settlement	t/ha/yr.	0.00	-	ArcGIS with NLWRA (2001b)

Feedlot data

At the feedlot, leaching from the feedlot pad was assumed to be negligible, because each of the facilities was designed and tested to strict standards in alignment with environmental regulation (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.

Each feedlot utilised land for effluent irrigation. This is used to grow forage crops which are then fed to cattle in the feedlot. The parameters used to determine sediment loss and nitrogen loss in runoff for the effluent irrigation area are shown in Table 47.

TABLE 47– PARAMETERS USED TO CALCULATE SEDIMENT LOSS AND NITROGEN LOSS IN RUNOFF FOR FEEDLOT EFFLUENT IRRIGATION LAND OCCUPATION

	Unit	Value	Uncertainty Range	Reference
Rainfall	mm	533-800	-	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 grain inputs for cattle production were based on a cropping process developed for the northern grain growing region (NSW north-east/west and QLD south-east/west regions). Soil and runoff losses were determined using six representative regions with the area. 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 from 2000-2011 was 595 mm with a range of 551-653 mm (BOM 2012).

Soil losses were based on estimates from the National Land and Water Resources Audit (NLWRA 2001b). 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 across 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 2001b) 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 48 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 48 – 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 (2001b)

Phosphorus depletion

P depletion was determined using a mass balance throughout the supply chain based on the law of the conservation of mass. 'Depletion' was defined as the loss of P from the production system to non-beneficial receptors. As an example, P lost in runoff or soil erosion to waterways was considered to contribute to P depletion, as this phosphorus is difficult to recover for agricultural use. In contrast, P in feedlot manure which is applied to agricultural land as a fertiliser was not considered a contributor to P depletion because this phosphorus is still cycling within the agricultural system. P taken up in livestock (LW) was considered a contributor to P depletion within the constraints of the system boundary for the study. However, when these data are extended to the meat processing and consumption phase, the flow of P will require further analysis. It is expected that P contained in LW flows to three main sources post slaughter; i) the meat processing waste stream, which is irrigated to land or emitted to water, ii) into animal by-products such as meat meal, which are predominantly used as a mono-gastric feed source, and iii) contained within the meat product which is consumed by humans, excreted and treated by municipal waste treatment processes.

Inputs

Livestock

The mass of livestock imported annually to each property was calculated from farm records. The chemical composition for beef entering and exiting the farms was estimated using BEEFBAL (McGahan et al. 2004) which assumes an average P concentration for store and finished beef cattle of 0.68% LW.

Feed and feed supplements

Significant amounts of nutrients can be brought on farm in feed purchased for livestock. The data supplied by producers in the survey for feed/feed supplement purchases were combined with standard figures for dry matter percentage and nutrient composition to estimate nutrient inputs.

Fertiliser

The mass of fertiliser applied on each supply chain property was collected from farm records. All values were calculated as property-scale averages. The composition (chemical analysis) of specific types of fertiliser can vary between manufacturers, but this is usually by less than 1% of P content. Common fertilisers were super phosphate (~ 9% P) for pastures and Di-Ammonium

Phosphate or DAP (20% P) for forage crops. The chemical analysis for the organic fertilisers and soil amendments used on the Victorian property were collected from the relevant manufacturer (Nutri-tech 2006).

Manure

In the grazing systems, excreted manure P was considered an embedded flow within the system. However, at the feedlot, manure is an export from the system in much the same way as cattle LW. Manure phosphorus from the feedlot is typically applied to surrounding crop and pasture land, where it represents a fertiliser replacement. Because this flow returns to crop or grazing land, it was not considered a contributor to P depletion from the system.

Outputs

Livestock

The mass of livestock exported annually from each property was calculated from farm records. Exported P was determined using the same values for imported livestock based on values from BeefBal.

Soil/Water loss

Losses of nutrients to the environment with eroded soil particles and overland flow can be a significant source of nutrient removal from the system. Methods for estimating P loss via this pathway are reported in the following section.

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, which 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 feedlots:

- Feedlot pen (drinking) water
- Other feedlot water uses – cattle washing, feed milling etc.
- Feedlot water supply system (Feedlot water balance 1)
- Feedlot runoff capture (Feedlot water balance 2)

Background system for farms and feedlots:

- 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. Background water use was reviewed to determine important processes (i.e. processes contributing >1%) and these processes were standardised to the methods used here where required. Methods and assumptions used to determine water use in each stage are provided in the following sections.

Impact assessment

The stress weighted water use impact assessment method applied different stress weighting factors for different regions of Australia where the farms and feedlots 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 feedlots are presented in Table 49.

TABLE 49 – SUMMARY OF SITE DATA USED IN WATER MODELLING FOR THE FOUR FARMS AND THREE FEEDLOTS

Farm	Rainfall (mm / yr.)	Evaporation (mm / yr.)	WSI
Nth NSW (Farm 1)	777	1463	0.011
Nth NSW (Farm 2)	661	1841	0.021
Sth NSW (Farm 3)	559	1863	0.815
SE VIC (Farm 4)	936	1165	0.012
Short-fed Feedlot	819	1717	0.021
Mid-fed Feedlot	524	1584	0.815
Long-fed Feedlot	819	1379	0.021

For background products that may be sourced from many regions, we applied the same approach as Ridoutt et al. (2012) 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. (2011) were applied in the present study. The feedlots under investigation in this study had metered records of water use from a previous study (Davis et al. 2008a).

Farm water supply balance

Water supplies were from creeks, bores, reticulated supplies or on-farm storage dams. Table 51 shows the different sources for water supply at each of 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. All 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 occupation change from the natural state into account. This is a major distinction. However, land occupation change was outside the scope of the study and accounting for this would affect multiple impact categories. However, we did conduct a preliminary assessment for one farm in order to gain an indication of the potential impact of including land occupation change into the assessment. Results are presented in Table 50, and show that changing land occupation from Eucalyptus forest to open pasture has the effect of generating water (represented by using negative values). In other words, beef production creates more water than it 'uses' because of the effect of land occupation change compared to the reference.

TABLE 50 – EFFECTS OF LAND OCCUPATION CHANGE ON THE WATER BALANCE OF A NORTHERN NSW BEEF FARM (FARM 1)

Land occupation	Runoff (% of rainfall)	Units	Annual Runoff
Estimate of change –low change (2% increase in runoff from forest to pasture, accounting for abstractions with farm dams)			
Reference system - Eucalyptus forest	5%	ML / yr.	137.1
Current land occupation - pasture with farm dams	7%	ML / yr.	184.1
Difference in water balance		ML / yr.	47.0
Difference in water use relative to beef production		L / kg LW	-1013
Estimate of change – moderate change values (5% increase in runoff from forest to pasture, accounting for abstractions with farm dams)			
Reference system - Eucalyptus forest	5%	ML / yr.	137.1
Current land occupation - pasture with farm dams	10%	ML / yr.	321.3
Difference in water balance		ML / yr.	184.1
Difference in water use relative to beef production		L / kg LW	-3970

Water sources

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 four farms was determined and is reported in Table 51.

TABLE 51 – SOURCES OF WATER SUPPLY FOR FARMS

Source of water supply	% of total water supply			
	Farm 1	Farm 2	Farm 3	Farm 4
Dam	67	83	25	100
Creek/River	33	17	45	0
Bore	0	0	30	0

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:

$$E = K_P \times E_{Pan} \qquad \textbf{Equation 1}$$

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 0.75 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 52. 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 52 – Evaporation and seepage supply efficiency factors

Inflows	Classification for impact assessment	Farm 1	Farm 2	Farm 3	Farm 4	Uncertainty (SD)
Net evaporation to drinking water supply ratio [a]	Consumptive use	0.88	2.47	1.68	-0.25	4.05
Seepage to drinking water supply ratio [b]	Transfer	1.49	1.46	1.20	0.97	4.05

[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.

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 feedlots multiplied by the average feed intake of cattle at each feedlot. 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 (Ridoutt et al. 2012).

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 the work by Ridoutt et al. (2012).

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. (2012). 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):

$$W_{evap\ loss} = C + (W_{total\ intake} - 3.3 \times DMI) \qquad \textbf{EQUATION 2}$$

Where:

C = 2.4 for calves, 3.4 for cows and bulls

Excreted urine and faeces (manure) water was determined by difference. For the feedlots this resulted in a manure moisture content of 87-93%.

Additional water use activities

Additional water use activities consisted of cleaning and minor water uses. Cleaning water use was made up of cattle washing, while minor uses included the trough cleaning water, evaporation from the troughs, and office and amenities water usage. These data were collected from measurements made at each feedlot 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 53 to Table 55 show the feedlot water supply balances for each feedlot.

TABLE 53 – FEEDLOT PEN WATER BALANCE FOR THE SF FEEDLOT

Source	Source Description	Use Description	Volume (L / finished animal)	Volume (ML)	Uncertainty (SD or range)
Inputs (source and use)					
Groundwater (stock)	Direct supply from bore (<500m depth),	Feedlot water supply (includes drinking water, losses, cleaning, maintenance)	3324.9	22.5	1.1
Feed (feed moisture and metabolic water)	Water taken up by plants, assumed to be green water source		522.3	3.5	1.43
Cattle (purchased animals brought to the farm)	Water accounted for in grazing processes		131.5	0.9	1.43
Total inputs			**3978.8**	**26.9**	
Outputs (source and use)					
Groundwater (stock)	Drinking water lost via the physiological processes of perspiration and respiration	Evaporative use	314.9	2.1	1.43
	Drinking water assimilated into the animal product	Catchment transfer	168.4	1.1	1.43
	Drinking water excreted in manure and urine	Evaporative use	3004.9	20.4	1.43
	Cattle washing	Evaporative use	0.0	0.0	1.1
	Minor uses	Evaporative use	490.6	3.3	1.1
Total outputs			**3978.8**	**26.9**	
Balance			**0.0**	**0.0**	

TABLE 54 – FEEDLOT PEN WATER SUPPLY BALANCE FOR MF FEEDLOT

Source	Source Description	Use Description	Volume (L / finished animal)	Volume (ML)	Uncertainty (SD or range)
Inputs (source and use)					
Groundwater (stock)	Direct supply from bore (<500m depth),	Feedlot water supply (includes drinking water, losses, cleaning, maintenance)	138.4	7.5	1.1
	Reticulated supply (town water)		5767.7	311.8	1.1
Feed (feed moisture and metabolic water)			898.6	48.6	1.43
Cattle (purchased animals brought to the farm)			153.5	8.3	1.43
Total inputs			6958.2	376.2	
Outputs (source and use)					
Groundwater (stock)	Drinking water lost via the physiological processes of perspiration and respiration	Evaporative use	903.4	48.8	1.43
	Drinking water assimilated into the animal product	Catchment transfer	223.9	12.1	1.43
	Drinking water excreted in manure and urine	Evaporative use	5220.1	282.2	1.43
	Cattle washing	Evaporative use	58.8	3.2	1.1
	Minor uses	Evaporative use	552.0	29.8	1.1
Total outputs			6958.2	**376.2**	
Balance			0.0	**0.0**	

TABLE 55 – FEEDLOT PEN WATER SUPPLY BALANCE FOR LF FEEDLOT

Source	Source Description	Use Description	Volume (L / finished animal)	Volume (ML)	Uncertainty (SD or range)
Inputs (source and use)					
River	Direct supply from river allocation	Feedlot water supply (includes drinking water, losses, cleaning, maintenance)	9368.0	274.0	1.1
Feed (feed moisture and metabolic water)			2358.4	69.0	1.43
Cattle (purchased animals brought to the farm)			163.0	4.8	1.43
Total inputs			11889.4	347.8	
Outputs (source and use)					
Groundwater (stock)	Drinking water lost via the physiological processes of perspiration and respiration	Evaporative use	2657.2	77.7	1.43
	Drinking water assimilated into the animal product	Catchment transfer	273.9	8.0	1.43
	Drinking water excreted in manure and urine	Evaporative use	7681.3	224.7	1.43
	Cattle washing	Evaporative use	1078.1	31.5	1.1
	Minor uses	Evaporative use	198.9	5.8	1.1
Total outputs			11889.4	**347.8**	
Balance			**0.0**	**0.0**	

Feedlot controlled drainage area water balance

Australian feedlots are designed to control drainage and overland flow around the feedlot site to restrict movement of manure nutrients to the environment. Within the controlled drainage area, runoff is greatly increased from hard surface areas (pens, roads). All water is captured in engineered effluent ponds, which are constructed with a storage capacity to limit effluent release to a one in ten year rainfall event. Excess water from the feedlot controlled drainage area either evaporates from the effluent pond, or is irrigated to grow crops. Because the feedlot site is highly modified, the most accurate way to determine the impact of the feedlot on the local hydrology was to compare the site to a reference, or 'green field' site (i.e. the feedlot site in the absence of the feedlot). We did this by modelling runoff from the greenfield site using USDA-SCS KII curve numbers (USDA-SCS 1972, USDA NRCS 2007). For the purposes of the water balance, we assumed the feedlot site released no water. Runoff from the reference, green field site was attributed to the feedlot as a water use. Data are shown in Table 56.

TABLE 56 – RUNOFF FROM REFERENCE LAND OCCUPATION ATTRIBUTED TO FEEDLOT CATTLE PRODUCTION AT THREE FEEDLOTS

	Units	SF Feedlot	MF Feedlot	LF Feedlot
Runoff from reference land occupation	ML/yr.	2.8	14.1	23.1
Runoff from feedlot controlled drainage area	ML / yr.	0.0	0.0	0.0
Consumptive water use attributed to cattle production	L / finished animal	413.4	260.2	790.4

Appendix 4 – Modelling GHG emissions

Enteric GHG emissions

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_{ij} – MJ CH_4/head/day) is as follows:

$$\boldsymbol{Y_{ij} = 3.406 + 0.510SR_{ij} + 1.736H_{ij} + 2.648C_{ij}} \qquad \textbf{EQUATION 3}$$

Where:

SR_{ij} = intake of soluble residue (kg/day)
H_{ij} = intake of hemicellulose (kg/day)
C_{ij} = 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.

The DCCEE provide default values for daily feed intake and feed properties for Australian feedlot cattle. However, for the three feedlots 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 feedlots relate to daily dry matter intake (DMI) and the proportion of grain, grass, legume and concentrate in the diets. Table 57, Table 58 and Table 59 show the daily feed intake and feed properties for the short, medium and long-fed feedlots used in this study.

TABLE 57 – DAILY FEED INTAKE AND FEED PROPERTIES FOR THE SF FEEDLOT

	Units	DCCEE (2010)	Actual data
Daily Intake (assume DMI)	kg/day	9.8	10.1
Proportion of grains in feed		0.779	0.796
Proportion of concentrates in feed		0.048	0.059
Proportion of grasses in feed[1]		0.138	0.130
Proportion of legumes in feed		0.035	0.014
Proportion of oil in feed			0.027
Enteric methane production – without accounting for oil	kg/hd/d	0.196	0.197
Enteric methane production – with Oil accounted for	kg/hd/d	n.a	0.167

[1] forage hay / silage classified under grasses

TABLE 58 – DAILY FEED INTAKE AND FEED PROPERTIES FOR THE MF FEEDLOT

	Units	DCCEE (2010)	Actual data
Daily Intake (assume DMI)	(kg/day)	11.7	10.6
Proportion of grains in feed		0.779	0.845
Proportion of concentrates in feed		0.048	0.100
Proportion of grasses in feed		0.138	0.004
Proportion of legumes in feed		0.035	0.051
Proportion of oil in feed			0.069
Enteric methane production – without accounting for oil	kg/hd/d	0.207	0.183
Enteric methane production – with Oil accounted for	kg/hd/d	n.a	0.113

TABLE 59 – DAILY FEED INTAKE AND FEED PROPERTIES FOR THE LF FEEDLOT

	Units	DCCEE (2010)	Actual data
Daily Intake (assume DMI)	(kg/day)	11.0	8.56
Proportion of grains in feed		0.779	0.684
Proportion of concentrates in feed		0.048	0.022
Proportion of grasses in feed		0.138	0.294
Proportion of legumes in feed		0.035	0.001
Proportion of oil in feed			0.001
Enteric methane production – without accounting for oil	kg/hd/d	0.213	0.195
Enteric methane production – with Oil accounted for	kg/hd/d	n.a	0.194

Grazing enteric methane

Enteric methane was modelled using the DCCEE (2010) methodology for pasture fed cattle in temperate Australia, which is based on Blaxter and Clapperton (1965). This approach requires the estimation of gross energy intake and then calculates the fraction 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. In order to determine the feed intake of the cattle, the equation derived by Minson and McDonald (1987) is used. This is then used to determine the gross energy intake and hence the enteric methane production.

In order to calculate feed intake (I_{ijkl} – kg dry matter/head/day) from live weight and live weight gain the following equation is used:

$$I_{ijkl} = (1.185 + 0.00454W_{ijkl} - 0.0000026W_{ijkl^2} + 0.315LWG_{ijkl})^2 \times MA_{ijkl=5}$$ **EQUATION 4**

Where:

W_{ijkl} = live weight in kg
LWG_{ijkl} = 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:

$$MA_{ijkl=5} = \left(LC_{ijkl=5} \times FA_{ijkl=5}\right) + ((1 - LC_{ijkl=5}) \times 1)$$ **EQUATION 5**

Where:

$LC_{ijkl=5}$ = proportion of cows>2 years old that are lactating
$FA_{ijkl=5}$ = feed adjustment (varies between 0 and 1.3 – see Table 6.B.5 (DCCEE 2010))

The gross energy content of feed dry matter is estimated to be 18.4 MJ/kg. Therefore, to determine the gross energy intake is found by multiplying the feed intake by this value:

$$GEI_{ijkl} = I_{ijkl} \times 18.4$$ **EQUATION 6**

The intake of the animals relative to that needed for maintenance is calculated using:

$$L_{ijkl} = I_{ijkl}/(1.185 + 0.00454W_{ijkl} - 0.0000026W_{ijkl^2} + (0.315 \times 0))^2$$ **EQUATION 7**

In order to determine the percentage of gross energy intake which yields enteric methane, the equation by Blaxter and Clapperton (1965) is used:

$$Y_{ijkl} = 1.3 + 0.112DMD_{ijkl} + L_{ijkl}(2.37 - 0.050DMD_{ijkl})$$ **EQUATION 8**

Where:

DMD_{ijkl} = digestibility of feed (%)
L_{ijkl} = feed intake relative to that needed for maintenance

Seasonal DMD values for pasture were based on the DCCEE (2010). Where these values did not align with cattle performance they were modified accordingly.

The methane yields (kg CH_4/head/day) for pasture fed cattle in temperate and tropical regions are then found using:

$$M_{temp} = \frac{Y_{ijkl}}{100} \times \frac{GEI_{ijkl}}{F}$$ **EQUATION 9**

Where:

F = 55.22 MJ/kg CH_4

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 11 shows a simplified mass balance for VS at Australian feedlots.

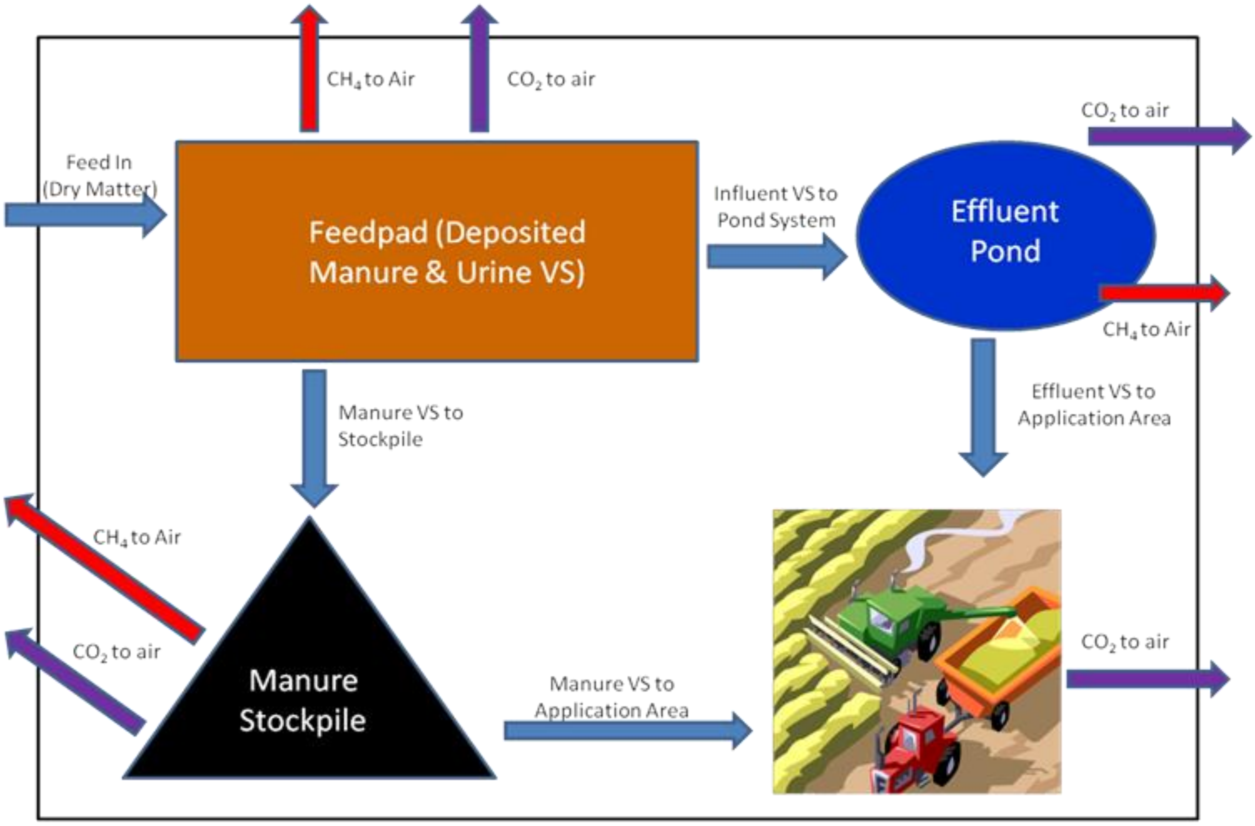

FIGURE 11 – THEORETICAL MASS BALANCE FOR EXCRETED VOLATILE SOLIDS IN AUSTRALIAN FEEDLOTS

The methane emission factors and ranges used for this study are summarised in Table 60.

TABLE 60 – 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 % 1.5%[1]	2.5-7.5% 0.75-2.25%	DCCEE 2010
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 between northern (5%) and southern (1.5%) regions

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 rations are shown in Table 61.

TABLE 61 –FEEDLOT DIETARY CRUDE PROTEIN LEVELS

Feedlot	Dietary CP (% dry basis)
Short fed	16.2
Mid fed	15.9
Long Fed	17.2

Feedpad emissions

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

$$Faecal_{MMS} = AF_{ijkl} \times MMS \times EF_{(MMS)} \times C_g$$ **EQUATION 10**

$$Urine_{MMS} = AU_{ijkl} \times MMS \times EF_{(MMS)} \times C_g$$ **EQUATION 11**

$$Total_{MMS} = Faecal_{MMS} + Urine_{MMS}$$ **EQUATION 12**

Where:

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

$EF_{(MMS)}$ = emissions factor (0.005 N_2O-N kg/N excreted – based on Muir (2011)) for the different manure management systems.

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

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 12 shows a generalised theoretical mass balance of nitrogen at a feedlot.

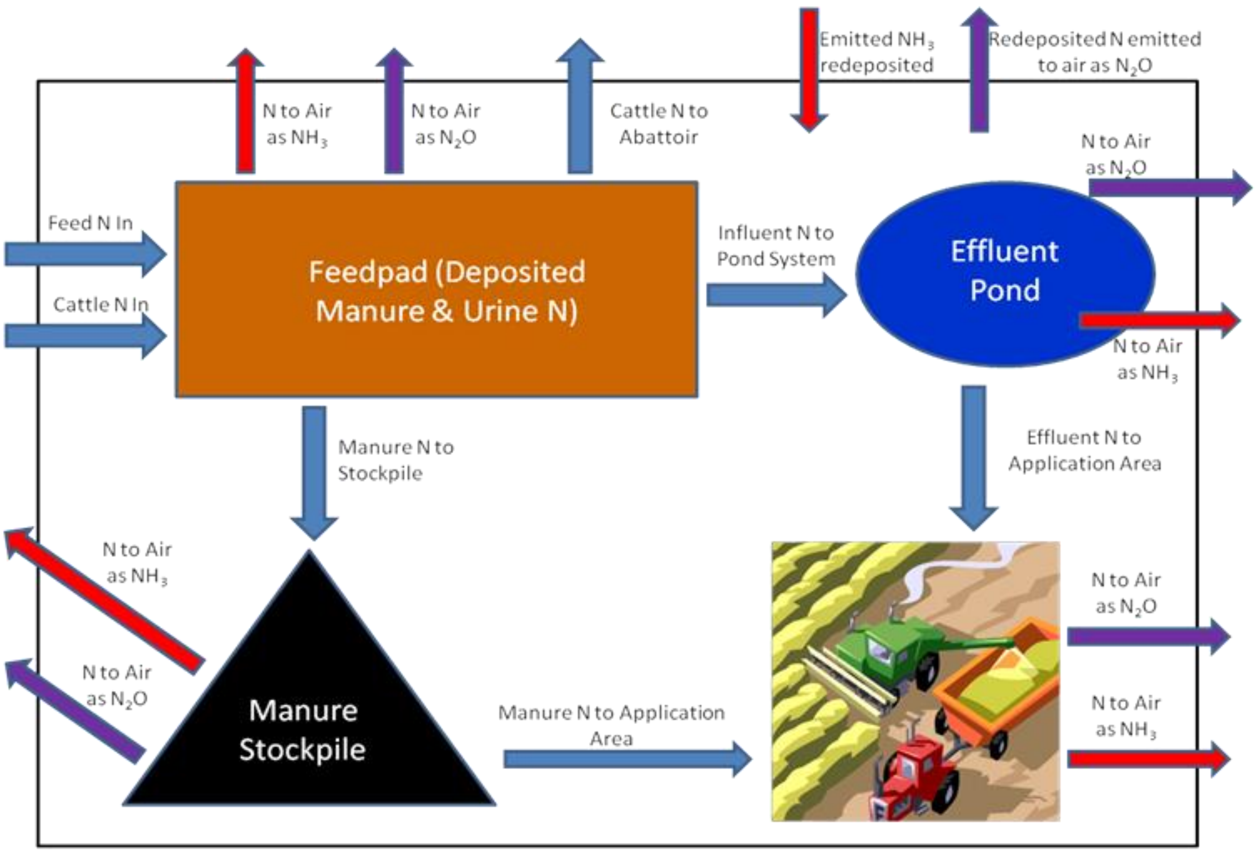

FIGURE 12 – THEORETICAL MASS BALANCE FOR EXCRETED NITROGEN IN AUSTRALIAN FEEDLOTS

Table 62 summarises the nitrous oxide and ammonia emission factors for feedlots used in this study.

TABLE 62 –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)

Grazing 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).

$$M = I \times (1 - DMD) \times MEF$$ **EQUATION 13**

Where:

M = methane yield (kg CH_4/head/day)
I = feed intake (kg dry matter/head/day) as calculated in Equation 3.
DMD = dry matter digestibility (%).
MEF = manure emission factor (0.000014 for temperate regions, and 0.000054 for tropical regions)

Grazing 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_{ijkl}) and amount of nitrogen retained by the body (NR_{ijkl}).

The crude protein intake CPI_{ijkl} (kg/head/day) of beef cattle is calculated using:

$$CPI_{ijkl} = I_{ijkl} \times CP_{ijkl} + (0.032 \times MC_{ijkl})$$ **EQUATION 14**

Where:
I_{ijkl} = dry matter intake (kg/head/day)
CP_{ijkl} = crude protein content of feed dry matter expressed as a fraction
MC_{ijkl} = milk intake (kg/head/day).

Nitrogen excreted in faeces (F_{ijkl} kg/head/day) is found using a similar method to that for feedlots (Section 0), however the contribution from milk protein is included in this case:

$$F_{ijkl} = \left\{0.3\left(CPI_{ijkl} \times \left(1 - \left[\frac{(DMD_{ijkl}+10)}{100}\right]\right)\right) + 0.105(ME_{ijkl} \times I_{ijkl} \times 0.008) + 0.08(0.032 \times MC_{ijkl}) + (0.0152 \times I_{ijkl})\right\}/6.25$$

EQUATION 15

Where:
DMD_{ijkl} = dry matter digestibility (expressed as a %)
ME_{ijkl} = metabolise energy (MJ/kg DM)
I_{ijkl} = feed intake (kg DM/head/day)
MC_{ijkl} = milk intake (kg/head/day)

Table 63 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 63 – DRY MATTER CRUDE PROTEIN (CP) CONTENT OF PASTURE

Farm	Av. CP (%)

Nth NSW – Farm 1	10
Nth NSW – Farm 2	9
Sth NSW – Farm 3	15
VIC – Farm 4	17

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

$$NR_{ijkl} = \{(0.032 \times MP_{ijkl}) + \{0.212 - 0.008(L_{ijkl} - 2) - [(0.140 - 0.008(L_{ijk_l} - 2)) / (1 + exp(-6(Z_{ijkl} - 0.4)))]\} \times (LWG_{ijkl} \times 0.92)\} / 6.25$$

Equation 16

Where:
MP_{ijkl} = milk production (kg/head/day)
L_{ijkl} = relative intake
Z_{ijkl} = relative size (live weight/standard reference weight)
LWG_{ijkl}= live weight gain (kg/day)

The amount of nitrogen excreted in urine is found using the equation XX from Section 0 for feedlot cattle. The nitrous oxide emissions from faecal and urinary nitrogen voided onto pasture are calculated using:

$$N_2O\ emissions = (AF_{ijkl} + AU_{ijkl}) \times MMS \times EF_{(MMS)} \times C_g$$

Equation 17

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.

Soil carbon flux potential

The estimation of 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, which was what this study has done.

Assessments of the potential carbon flux from soil under improved pastures were carried out for all the farms. The purpose of including a soil carbon change scenario was to investigate the relative significance of soil carbon sequestration to net GHG emissions at carbon sequestration rates found in published Australian studies. Carbon sequestration in soil is subject to a high degree of uncertainty and may have a large impact on net GHG. Across the globe, soil carbon levels have often been found to increase under pastures when improved species and fertiliser are added (Conant et al. 2001). However, soil carbon sequestration under improved pastures (phosphorus fertiliser and introduced legumes) compared with native pastures in Australia has been shown to vary from zero (Wilson et al. 2010) to between 0.26 and 0.72 t C/ha.yr (Chan et al. 2010). Chan et al (2010) observed average soil carbon sequestration rates of 0.41 t C/ha.yr (assumed over a 40 year time period) from pastures fertilised with phosphorus and the concurrent introduction of legumes. For the present study, the soil carbon flux scenario investigated carbon sequestration at a linear rate of at a linear rate of 0.41 t C ha.yr for either 25 years or 40 years (Chan et al. 2010) before establishing a new equilibrium. These two scenarios resulted in a calculated total soil carbon increase of 10.25 t C ha (25 year sequestration period) or 16.5 t C ha (40 year sequestration period). These were divided by 100 to provide an annual sequestration rate over a 100 year timeframe.

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

$$Soil_{C\,Loss} = \frac{0.1756 \times 100}{Clay\,content}$$

Equation 18

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 64 and Table 65, along with the assumed uncertainty.

TABLE 64 – KEY GHG PARAMETERS USED FOR GRAZING CATTLE WITH UNCERTAINTY

Emission source	Key parameters / model	Assumed Uncertainty	Reference
Enteric methane (temperate climate)	M (kg/hd) = (Y (% Gross Energy Intake as CH_4) / 100) x (GEI (MJ/kg) / F (MJ / kg CH_4)	± 20%	DCCEE (2010) – from Blaxter and Clapperton (1965)
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)
Legume pasture emissions	0.0125 kg N_2O-N / kg N residue deposited to soil from trampling and senescence	± 50%	DCCEE (2010)

TABLE 65 – 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	Faecal and urinary 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)