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Scoping Life Cycle Assessment of the Australian lot feeding sector

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Abstract

Life cycle assessment (LCA) is a comprehensive, supply chain environmental assessment tool that investigates environmental impacts for a product, such as a kilogram of beef. LCA research has previously been conducted in the Australian beef industry by Peters et al. (2009a, b, 2010), who modelled data collected by Davis & Watts (2006). This project built on previous LCA research in the industry, with particular attention being given to modelling feedlot emissions and impacts. Impacts assessed included global warming potential (GWP), water usage and primary energy (PE) usage. The primary functional unit applied was '1 kg of liveweight (LWT) gain at the feedlot from point of induction to immediately prior to transport for slaughter', representing a 'gate-to-gate' assessment of the feedlot. Upstream and downstream processes were also modelled to contextualise the results.

Feedlot gate to gate GWP ranged from 7.5 kg CO_2-e / kg LWT gain to 11.3 kg CO_2-e / kg LWT gain, and was dominated by enteric methane emissions. 'Blue' water usage at the feedlot ranged from 151 to 871 L / kg LWT gain. Water usage at the feedlot was dominated by water carried through with irrigated commodities (up to 95% of water usage). Primary energy usage ranged from 34.5 to 49.1 MJ / kg LWT gain. Energy embedded within feed contributed 89 – 90% of total energy usage at the feedlot. GWP, water and energy usage were considerably higher than previously estimated by Davis & Watts (2006) because of the broader scope of this research.

Executive Summary

Life cycle assessment (LCA) is a comprehensive, supply chain environmental assessment tool that investigates environmental impacts for a product, such as a kilogram of beef. LCA research has previously been conducted in the Australian beef industry by Peters et al. (2009a, b, 2010), who modelled data collected by Davis & Watts (2006). Davis & Watts (2006) and later Davis et al. (2008) have developed comprehensive datasets for livestock performance, energy and water usage at Australian feedlots, though to date these data have only been integrated into one beef supply chain (Peters et al. 2009a, 2010). This project built on previous LCA research in the industry, with particular attention being given to modelling feedlot emissions and impacts. The study investigated two supply chains, with attention being focussed on the feedlot stage. Both supply chains used generic upstream and downstream processes (such as cattle breeding and meat processing) to highlight the differences between the lot-feeding systems.

Impacts assessed included global warming potential (GWP), water usage and energy usage at the feedlot in a gate to gate study. The primary functional unit applied was ‘1 kg of liveweight (LWT) gain at the feedlot from point of induction to immediately prior to transport for slaughter’.

The study focused on two very different feedlots (one smaller feedlot feeding for domestic markets and one larger feedlot feeding for long-fed export markets). These two feedlots could not be considered representative of the whole industry. As the results are preliminary in nature, they should not be considered as industry averages without further research.

Global Warming Potential

Feedlot gate to gate GHG emissions were also investigated. For supply chain 1 (short fed), emissions were 7.5 kg CO_2-e / kg LWT gain and for supply chain 2 (long fed) emissions were 11.3 kg CO_2-e / kg LWT gain. The major contributions to emissions at the feedlot were enteric methane (about 40 – 45%), ration production (about 25 – 30%) and feedpad emissions (about 20%). All other contributions (energy usage, manure storage, treatment and reuse etc) amounted to approximately 10%. The higher emissions for the long fed supply chain are primarily driven by the lower production efficiency (feed conversion) of very long fed cattle (> 300 days) compared to short fed cattle (70 days). It is noted that assessment of feed grains and other commodities relied on a series of desktop studies. Considering their importance at the feedlot, further research in this area is warranted to ensure correct data are used.

From an analysis of the manure management system at the feedlots the results from the theoretical mass balance show emissions of 1.98 kg CO_2-e / kg LWT gain for short fed cattle and 3.09 kg CO_2-e / kg LWT gain for long fed cattle. An analysis was also conducted for the short fed scenario using the Department of Climate Change (DCC) methodology and manure management system emissions contributed 2.15 kg CO_2-e / kg LWT gain, or 8.6% higher than the theoretical mass balance approach.

This study provided a more comprehensive assessment of manure GHG than previous feedlot research, such as Davis & Watts (2006). However, the accuracy of the findings is limited by the available research into nitrogen and volatile solids flows at feedlots, and specific emissions factors. To date, there are large gaps in the knowledge of these issues for Australian feedlots.

This project has identified the key emission factors for the manure stream, allowing prioritisation of R&D needs for the industry. These are summarised in Table 1.

TABLE 1 – KEY EMISSION FACTORS AND RESEARCH NEEDS FOR ESTIMATING MANURE GHG

Emission source	Assessment method	Contribution to Manure GHG	R&D Ranking
Storage and Feedpad N_2O	Mass balance / DCC	62 – 72%	1
Atmospheric deposition (N_2O)	DCC	9%	2
Ammonia volatilisation (NH_3)	Mass balance	indirect	2
Feedpad CH_4	Mass balance / DCC	3 – 10%	3*
Manure Application (N_2O)	Mass balance / DCC	21%	4

* Feedpad methane may be investigated concurrently with feedpad nitrous oxide.

The interrelationship between ammonia emissions and nitrous oxide from manure application and atmospheric deposition highlight the importance of a mass balance approach to research in this area. Mass balance theory is essential to the DCC method of approach (as emission factors are all related back to nitrogen intake and excretion. Considering this, research in this area needs to use an integrated mass balance approach to ensure accurate results and emission factors are generated. Although not studied in this LCA, different nitrogen intake levels will also influence manure GHG, possibly to a large extent.

When the full supply chain was considered for context, GWP was estimated to be 16 kg CO_2-e / kg HSCW for both the domestic short fed and export long fed supply chains. These results are in the range but at the higher end of comparable international studies, and are close to 50% higher than the results reported by Peters et al. (2010). This is due to several cumulative factors, including the use of a Queensland breeding system which has higher enteric methane emissions (calculated with the DCC methodology, compared to the southern states), higher feedlot emissions, more inclusive commodity production emissions at the feedlot and more inclusive modelling of GHG emissions from livestock at the breeder farm. It should be noted that this was a scoping study, and did not include some minor emission sources. Consequently total GHG burdens may be higher than this. However, sequestration options and other offsets (such as substitution of manure for fertiliser) will reduce overall emissions. These options will be studied in more detail in further LCA research.

Water and Energy Usage

At the feedlot, primary energy (PE) usage ranged from 34.5 to 49.1 MJ / kg LWT gain for the supply chain 1 (short fed) and supply chain 2 (long fed) respectively. Energy embedded within feed contributed 89 – 90% of total energy at the feedlot, which included minor contributions from transport and fuel use at the feedlot for feed delivery. Energy usage was considerably higher than previously estimated by Davis & Watts (2006) and Davis et al. (2008) because of the broader scope of this research (which included upstream energy usage associated with grain).

The study was based on a series of desktop studies for feed grains and forage. Considering their importance, further research in this area is warranted to ensure correct data are used.

When the full supply chain was considered for context, PE was found to range from 14.3 MJ / kg HSCW for supply chain 1 (short fed) to 32.2 MJ / kg HSCW for supply chain 2 (long fed). These results were generally comparable to previous Australian research and the literature.

Blue water usage was 871 L / kg LWT gain for feedlot 1 (short fed) and 151 L / kg LWT gain at feedlot 2 (long fed). Water usage at the feedlot was dominated by water carried through with irrigated commodities (95% of the water usage for feedlot 1). Direct water usage at the feedlot was a smaller contributor. These results are considerably higher than previously estimated because of the inclusion of water associated with irrigated commodities and water captured at the feedlot in effluent holding ponds. Though subject to debate, the inclusion of water from these sources is in line with recent advances in LCA and water footprinting research.

When the full supply chain was considered for context, blue water usage was 460 L / kg HSCW for supply chain 1 (short fed) and 222 L / kg HSCW for supply chain 2 (long fed). These values were within the range suggested by Peters et al. (2009a). When blue and green water were combined (comparable to the virtual water or water footprint of beef) the water usage was 21,606 L / kg HSCW for supply chain 1 (short fed) and 18,612 L / kg HSCW for supply chain 2 (long fed).

Previous research (Peters et al. 2009a) suggested water usage for beef production was as low as 27 L / kg HSCW, which is clearly well below the drinking water requirements of a beef herd. This was because the method applied did not require accounting of water that is not ‘pumped’, thereby excluding water captured in farm dams for example. As highlighted by Wiedemann et al. (2010a), water usage is likely to be higher than previously estimated by Peters et al. (2009a) because of the continued use of irrigation for feeding cattle, and from the use of irrigated commodities as supplements for feedlot rations. Further research is needed to gain more representative data on this however.

Detailed research recommendations are supplied in the report.

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List of Abbreviations

ABARE – Australian Bureau of Agricultural and Resource Economics

ABS – Australian Bureau of Statistics

CFCs – Chlorofluorocarbons

CH_4 – Methane

CO_2 – Carbon Dioxide

COD – Chemical Oxygen Demand

CPRS – Carbon Pollution Reduction Scheme

DCC – Department of Climate Change

ECM – Energy Corrected (raw) Milk

EREP – Environmental Resource Efficiency Plan

ETS – Emission Trading Scheme

GHG – Greenhouse Gas

GHGP – Greenhouse Gas Protocol Initiative

GM – Genetically Modified

GWP – Global Warming Potential

HFCs – Hydrofluorocarbons

HSCW – Hot Standard Carcass Weight

IO-LCA – Input-Output Life Cycle Assessment

IPCC – Intergovernmental Panel on Climate Change

LCA – Life Cycle Assessment

LCI – Life Cycle Inventory

LPG – Liquid Petroleum Gas

MLA – Meat & Livestock Australia

MRET – Mandatory Renewable Energy Targets

N_2O – Nitrous Oxide

NCAS – National Carbon Accounting System

NGERS – The National Greenhouse and Energy Reporting System

NGA – National Greenhouse Accounts

NGGI – National Greenhouse Gas Inventory

O_3 – Ozone

OA – Organic Agriculture

PE – Primary Energy

PFCs – Perfluorocarbons

REC's – Renewable Energy Certificates

SFs – Sulphur hexafluoride

UNFCCC – United Nations Framework Convention on Climate Change

VW – Virtual Water

WRI – World Resources Institute

WUE – Water Use Efficiency

1 Introduction

1.1 Project objectives and reporting

It is becoming increasingly obvious that life cycle assessment (LCA) is the best tool to develop the "carbon footprints" of various agricultural commodities. However, there is currently insufficient quality inventory data available in Australia. Many good-quality research studies cannot be used because they do not link their research results to some functional unit (e.g. kg of liveweight gain). It is proposed to develop an LCA model for the Australian feedlot sector in two stages.

- Stage 1 – Rapid LCA for model development, sensitivity testing and R&D guidance (this Project).
- Stage 2 – Detailed LCA for incorporation into whole-supply-chain red-meat LCA models (later work).

The general LCA process is outlined in a series of Australian and international standards and will be adhered to in the goal and scope development process. In addition to this, the Methodology for Agricultural Life Cycle Assessment in Australia developed for the Rural Industries Research and Development Corporation (RIRDC) (Harris and Narayanaswamy, 2009) will be used in this project where this is agreed to be 'state-of-the-art'.

The objectives of this project are to:

1. Develop a complete model of the whole feedlot system.
2. Produce a first-order estimate of the resource usage (energy and water) and greenhouse gas (GHG) emissions per kg liveweight (LWT) gain in the feedlot sector including the feed grain component.
3. Complete a sensitivity analysis of the major parameters.
4. Determine the relative impact of different markets (short fed vs long fed) on resource usage and impacts per kg LWT gain at the feedlot.
5. Prioritise subsequent research and data collection efforts to those areas where the emissions are highest per kg LWT gain.
6. Provide guidance on best-bet greenhouse gas (GHG) mitigation options if these are required by industry in the short term.

1.2 Research background

1.2.1 Greenhouse gas, energy and water research

Meat & Livestock Australia (MLA) have commissioned several projects to investigate the environmental performance of Australian feedlot beef production with respect to GHG emissions, energy and water usage. 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.

Research into GHG emissions at the feedlot has progressed in a number of areas including nitrous oxide emissions, methane emissions and enteric methane. Livestock GHG research tends to be highly specific, focussing on rumen manipulation or soil management for example. Few projects attempt to contextualise research findings or to examine the trade-offs that may exist with mitigation strategies. This is particularly relevant where an environmental impact may bridge fields of science that are not generally related, such as animal nutrition and soil science (of great relevance to livestock related nitrous oxide emissions for example).

LCA is a very useful tool for drawing these research areas together, quantifying emission areas and mitigation potential and providing results in the context of beef production.

Another key issue to the industry is the estimation of water usage for red meat production. While a great deal of work has been completed on water usage in the prior MLA funded project B.FLT.339, this did not account for ‘upstream’ water usage such as that associated with feed production. This will be an important component of the ‘water footprint’ for feedlot beef.

This project follows on from several projects previously 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 data for the modelling. Rather than replicating these projects, an outline of key projects and reports is supplied in this section and will be referred to where relevant in the report.

B.CCH.2022 – Review of water use and GHG emissions from red meat production – Commissioned February 2009 and completed August 2009 (led by FSA Consulting).

This project was presented as three reports:

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

LCA theory, extensive background literature and context are available in this report and have been summarised where relevant. The report is referenced as Wiedemann et al. (2010a).

Report 2 – Enteric Methane Review is focussed on this issue alone because of the significance of this emission source to the red meat industries. The report was compiled by Dr David Cottle and Professor John Nolan from the University of New England (UNE), covering nutritional and genetic approaches to mitigation of emissions from livestock, modelling of livestock emissions and a review of the Department of Climate Change (DCC) methodologies available for the red meat industries.

Data from this report were used for estimating enteric methane emissions in this LCA project. This report is referenced as Cottle & Nolan (2010).

Report 3 – Nitrous Oxide and Carbon Cycling in Soils and Waste Review – This report was compiled by Dr Matt Redding, and covers all emissions related to nitrous oxide and carbon (i.e. non enteric methane) from across the red meat supply chain, with particular attention to the feedlot sector.

Data from this report were used for sensitivity analysis and construction of a theoretical mass balance in this LCA project. This project is referenced as Redding (2010).

B.FLT.0339 – Water and energy usage for individual activities within Australian feedlots – current project (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 the input for all modelling completed in this LCA, and data collection methods will not be replicated in this report. This project is referenced as Davis et al. (2008).

2 Greenhouse gas background

2.1 The Greenhouse Effect

Despite the widespread use of the terms ‘global warming’ and ‘the greenhouse effect’, many people do not have a clear understanding of the fundamental processes that drive these effects. These processes are summarised here.

The earth is surrounded by an atmosphere that protects it from high-energy radiation and absorbs heat to provide a moderate climate that supports life. The earth’s atmosphere behaves like the roof of a greenhouse, allowing short-wavelength solar radiation from the sun, predominantly in the visible or near visible (e.g. ultraviolet) part of the spectrum to pass through it and warm up the surface of the earth. Roughly one-third of the solar energy that reaches the top of Earth’s atmosphere is reflected directly back into space. The remaining two-thirds are absorbed by the surface and, to a lesser extent, by the atmosphere. The reflected thermal radiation is re-radiated from the earth’s surface at much longer wavelengths, primarily in the infrared part of the spectrum. Much of this thermal radiation emitted by the land and ocean is absorbed by gases in the atmosphere that are opaque to infra-red radiation, and is re-radiated back to Earth. This capture of thermal radiation is called the greenhouse effect, and the gases that absorb the emitted heat are known as greenhouse gases (Le Treut et al. 2007). The greenhouse effect is a natural phenomenon that is essential to life on earth, however since the industrial revolution there has been an increase in greenhouse gas emissions and hence greenhouse gas concentrations from human activity (anthropogenic greenhouse gases). Figure 1 shows an idealised model of the greenhouse effect on energy radiated from the earth.

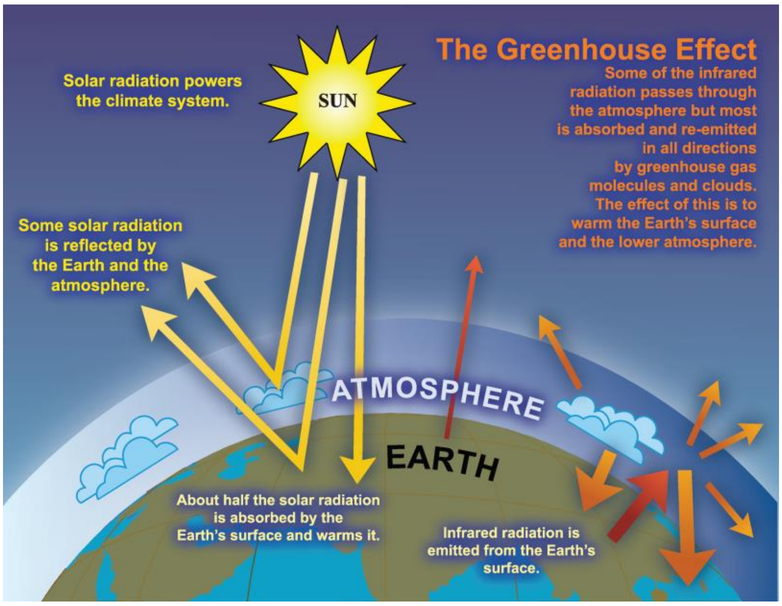

FIGURE 1 – AN IDEALISED MODEL OF THE NATURAL GREENHOUSE EFFECT (LE TREUT ET AL. 2007)

Figure 2 is a schematic representation of the flows of energy between outer space, the Earth's atmosphere, and the Earth’s surface. This shows how these flows combine to trap heat near the surface and create the greenhouse effect. The ability of the atmosphere to capture and recycle energy emitted by the Earth’s surface is the defining characteristic of the greenhouse effect. To use a greenhouse as an example, the glass walls reduce airflow and increase the temperature of the air inside. Analogously, but through a different physical process, the Earth’s greenhouse effect warms the surface of the planet. Without the natural greenhouse effect, the average temperature at Earth’s surface would be below the freezing point of water as all energy would be lost to outer space. However, too much radiation capture means that the earth begins to heat up. Hence, the balance between the energy entering and leaving the system is what determines whether the earth gets warmer, cooler or stays the same.

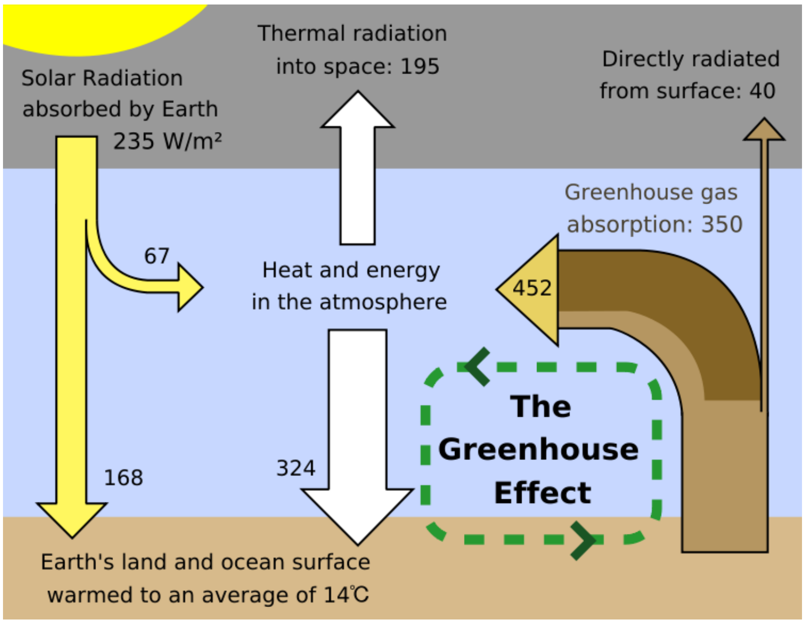

FIGURE 2 – THE RADIATION (ENERGY) BALANCE OF THE EARTH (ROHDE 2008).

2.2 Greenhouse gases (GHG)

The two most abundant gases in the atmosphere, nitrogen (comprising 78% of the dry atmosphere) and oxygen (comprising 21%), exert almost no greenhouse effect. Instead, the greenhouse effect comes from molecules that are more complex and much less common (Le Treut et al. 2007). The gases with the greatest influence on global warming are water vapour (H_2O), carbon dioxide (CO_2), nitrous oxide (N_2O), methane (CH_4) and ozone (O_3). In addition, there are a range of human-made halocarbons (such as perfluorocarbons (PFCs), hydrofluorocarbons (HFCs), chlorofluorocarbons (CFCs) and sulphur hexafluoride (SF_6) that exist in small amounts but are very potent and contribute to the total warming (Garnaut 2008). Compared to nitrogen and oxygen, which collectively comprise 99 per cent of the volume of the atmosphere, greenhouse gases occur only at trace levels, making up just 0.1 per cent of the atmosphere by volume (IPCC 2001a). Despite the low concentration of greenhouse gases in the earth's atmosphere, their presence means that the earth has an average global surface temperature of about 14ºC—about 33ºC warmer than if there were no greenhouse gases at all (Intergovernmental Panel on Climate Change – IPCC 2007).

Only some of these gases are directly emitted by human activities. Humans have less direct control over gases such as water vapour and ozone, although concentrations of these gases can be affected by human emissions of other reactive gases (Garnaut 2008).

After water vapour, carbon dioxide is the most abundant greenhouse gas in the atmosphere. Most gases are removed from the atmosphere by chemical reaction or are destroyed by ultraviolet radiation. Carbon dioxide, however, is very stable in the atmosphere. Hence, this leads to the whole discussion about "carbon". However, there are many other GHG's and some of these do not include any carbon, e.g. N_2O and SF_6, hence carbon is somewhat of a misnomer.

The warming of the atmosphere by different greenhouse gases is compared using the global warming potential (GWP). This compares the radiative forcing from a given mass of greenhouse gas to the radiative forcing caused by the same mass of carbon dioxide and is evaluated for a specific timescale (CASPI 2007). GWP depends both on the intrinsic capability of a molecule to absorb heat, and the lifetime of the gas in the atmosphere. The GWP values take into account the lifetime, existing concentration and warming potential of gases. Thus, GWP values will vary depending on the time period used in the calculation (Garnaut 2008). If a molecule has a high GWP on a short time scale (say 20 years) but has only a short lifetime, it will have a large GWP on a 20-year scale but a small one on a 100-year scale. Conversely, if a molecule has a longer atmospheric lifetime than CO_2, its GWP will increase with time. For example, sulphur hexafluoride has the highest GWP of all gases at 22,800 times that of carbon dioxide because it has a long atmospheric lifetime of 3200 years, but has a low impact on overall warming due to its low concentrations.

GWP is used under the Kyoto Protocol to compare the magnitude of emissions and removals of different greenhouse gases from the atmosphere. The Kyoto Protocol establishes legally binding commitments for the reduction of four greenhouse gases (carbon dioxide, methane, nitrous oxide, sulphur hexafluoride), and two groups of gases (hydrofluorocarbons and perfluorocarbons).

The GWP of the four greenhouse gases and two groups of gases (HFCs and PFCs) is shown in Table 2. The GWP of each greenhouse gas is expressed on a carbon dioxide equivalency (CO_2-e) basis. Contributing greenhouse gases are multiplied by their GWP to determine an equivalent amount of emitted CO_2. Carbon dioxide equivalency is a quantity that describes, for a given mixture and amount of greenhouse gas, the amount of CO_2 that would have the same GWP, when measured over a specified timescale (generally 100 years).

TABLE 2 – THE GLOBAL WARMING POTENTIAL OF THE MAJOR GREENHOUSE GASES

Greenhouse Gas	Lifetime in the atmosphere (years)	100 year global warming potential
Carbon Dioxide	Variable	1
Methane	12	25
Nitrous Oxide	114	298
Sulphur hexafluoride	3200	22800
HFCs	1.4 - 270	124 - 14800
PFCs	740 – 50,000	7400 - 17700

Source: IPCC (Solomon et al. 2007).

Two compounds of particular importance to the carbon emissions from red meat production are methane and nitrous oxide. Methane (CH_4) has a GWP 25 times that of CO_2 while nitrous oxide (N_2O) has a GWP 298 times that of CO_2 when measured on a 100 year timescale. It is noted that the potentials reported in Table 2 vary depending on source, and may be slightly different in other sections of the project reports depending on the framework under which the research is being considered.

2.3 Red meat industry GHG emissions

Red meat production has a number of potential sources of GHG emissions. Of these, enteric methane is the most significant. Enteric methane is a by-product of the fermentation processes in the gut of a ruminant (to be discussed in detail later in this report). However, depending on the way emissions are accounted, there is a wide range of GHG emissions that could be attributed to red meat production. Table 3 gives a summary of possible GHG emissions broken down by type, sector in the supply chain and scope (definition of scope 1, 2 and 3 emissions is provided in section 3.2.1).

TABLE 3 – EXAMPLES OF GHG EMISSIONS FOR THE RED MEAT INDUSTRY

	Grazing Sector	Feedlot Sector	Processing Sector	Distribution Sector

Scope 1	Enteric emissions from livestock (CH_4) Manure management emissions (CH_4, N_2O) Land use emissions (primarily N_2O) Fuel Usage on-farm (CO_2)	Enteric emissions from livestock (CH_4) Manure management emissions (CH_4, N_2O) Fuel Usage on-site (CO_2)	Emissions from waste treatment ponds (CH_4, N_2O) Hydrofluorocarbon (HFC) emissions during the use of refrigeration equipment Fuel Usage on-site (CO_2)	Hydrofluorocarbon (HFC) emissions during the use of refrigeration equipment Fuel Usage on-site (CO_2)
Scope 2	On-farm electricity use	Feedlot electricity Use	Abattoir Electricity Use	Electricity use in cooling product
Scope 3	Agricultural and veterinary chemicals Off-farm fuel usage for livestock transport Embedded energy in plant and infrastructure	Feed grains and fodders Agricultural and veterinary chemicals Off-farm fuel usage for livestock / commodity transport Embedded energy in plant and infrastructure	Packaging Off-site transport fuel Embedded energy in plant and infrastructure	Packaging Off-site transport fuels Embedded energy in plant and infrastructure

3 Greenhouse gas accounting methods

Several mechanisms have been implemented or suggested as a means of mitigating GHG production within the context of climate change or reducing atmospheric pollution. Examples include emission trading schemes (carbon trading) and international agreements such as the United Nations Kyoto Protocol. These activities require the ability to accurately measure GHG emissions and sinks. GHG accounting is a complex process that needs to encompass both emissions and sinks (sequestration) of GHG's over a specified time span within a physical or business boundary. There are a number of methodologies which provide a framework for the estimation of GHG emissions. The following sections will serve as a platform for the red meat industry to discuss the approaches and to find a way forward on how it can best profit from information gained from each of these methodologies.

3.1 National greenhouse gas inventories

In 1997, the Kyoto Protocol was adopted following a meeting of all major countries in Kyoto, Japan. The objective is to achieve stabilisation of GHG concentrations in the atmosphere at a level that would prevent dangerous anthropogenic (man-made) interference with the climate system.

The Kyoto Protocol is an agreement made under the United Nations Framework Convention on Climate Change (UNFCCC). Countries that ratify this protocol commit to reducing their emissions of CO_2 and the five other GHG's, or to engage in emissions trading if they maintain or increase emissions. The Kyoto Protocol now covers 181 countries globally but only 60% of countries in terms of global GHG emissions. As of December 2007, the USA and Kazakhstan are the only signatory nations to have signed but not ratified the act. The first commitment period of the Kyoto Protocol ends on December 31, 2012, and international talks began in May 2007 on a subsequent commitment period.

Under the Kyoto Protocol, negotiations occurred that allowed different countries to have different reductions (or increases) in GHG emissions. National limitations range from 8% reductions for the European Union and some others to 7% for the US, 6% for Japan, 0% for Russia, and permitted increases of 8% for Australia and 10% for Iceland.

For the Kyoto Protocol to be monitored, it is necessary to calculate the GHG emissions for individual countries for individual years from 1990 onwards. A National Greenhouse Gas Inventory (NGGI) is the total GHG emissions from a country over a year. It is immediately evident that a standard GHG accounting procedure must be developed so that all countries report their emissions fairly and equitably.

The IPCC is a scientific body tasked to evaluate the risk of climate change caused by human activity. The panel was established in 1988. In 1996, IPCC came up with a methodology for nations to use to calculate their NGGI. In 2000, IPCC presented a Good Practice Guideline for preparing a NGGI. When calculating a NGGI, a nation can use IPCC default methods or develop country-specific methods and factors (for larger, more important emissions).

Australia conducts a NGGI each year. The DCC provides methodologies for the calculation of GHG emissions for each sector (http://climatechange.gov.au/inventory/methodology/index.html) (DCC 2007b). The most recent methodology for agriculture was published in 2009. Factors and methods for the estimation of individual emissions (i.e. enteric methane) can be drawn from the NGGI methodology for use at an industry or individual enterprise level.

In 2007, it was calculated that agriculture produced 88.1 Mt Co_2-e or 16.3% of Australia's GHG emissions, making it the second largest emitting sector behind stationary energy (DCC 2009b, see figure 3). This contribution rises to 23% when the energy and transport used by the agricultural sector is included. The *Agriculture* sector is the dominant national source of both methane and nitrous oxide – accounting for 67.9 Mt CO_2-e (58.9%) and 20.2 Mt CO_2-e (85.9%) respectively of the net national emissions for these two gases.

GHG emissions from *Agriculture* increased by 1.5% (1.3 Mt) between 1990 and 2007, and decreased by 3.0% (2.7 Mt) from 2006 to 2007. Preliminary estimates for 2008 indicate that Agriculture emissions have increased by 3.0% (2.6 Mt) since 2007 due to increased emissions from savanna burning (DCC 2009b).

Sector and Subsector	Emissions Mt CO_2-e				
	CO_2	CH_4	N_2O	HFCs/PFCs/SF_6	Total
All energy (combustion + fugitive)	372.1	33.3	2.7	NA	408.2
Stationary energy	289.5	1.3	1.0	NA	291.7
Transport	76.5	0.6	1.7	NA	78.8
Fugitive emissions	6.2	31.5	0.0	NA	37.7
Industrial Processes	24.1[a]	0.1	0.0	6.1	30.3
Agriculture	NA	67.9	20.2	NA	88.1
Waste	0.0	13.9	0.6	NA	14.6
National Inventory	**396.3**	**115.3**	**23.5**	**6.1**	**541.2**

FIGURE 3 – GHG EMISSIONS BY SECTOR IN AUSTRALIA IN 2007 (DCC 2009B)

Greenhouse gas source and sink categories	CO_2-e emissions (Gg)			
	CO_2	CH_4	N_2O	Total
4 AGRICULTURE	**NA**	**67950**	**20156**	**88106**
A Enteric fermentation	NA	57561	NA	57561
B Manure management	NA	1859	1594	3453
C Rice cultivation	NA	196	NA	196
D Agricultural soils	NA	NA	15002	15002
E Prescribed burning of savannas	NA	8122	3463	11585
F Field burning of agricultural residues	NA	211	98	309

FIGURE 4 – GHG EMISSIONS FROM AGRICULTURE IN AUSTRALIA IN 2007 (DCC 2009B)

Figure 4 illustrates 2007 data and shows that enteric emissions are the largest component of agriculture's emissions followed by agricultural soils (mainly N_2O emissions from fertiliser usage). Manure management (4%) is the estimation of GHG emissions from manure primarily in the intensive livestock industries (lot feeding, pigs, poultry and dairy).

However, Australia's methane impact is further understated because the DCC uses a GWP for methane of 21 and not 25 for their greenhouse gas inventory calculations (Australian Greenhouse Office 2006a; DCC 2008e). This is due to the UNFCCC having agreed that the revised figures of GWP for different gases will not apply to greenhouse gas reporting until the second commitment period (2013-2017). This has serious implications for livestock methane emissions. Similarly, for N_2O emissions the DCC uses a GWP of 310 and not 298 for their greenhouse gas inventory calculations (Australian Greenhouse Office 2007a; DCC 2008e).

Because the NGGI relies on an industry-by-industry approach to calculate emissions, it is not comparable to other forms of accounting such as carbon footprinting or LCA.

3.2 Carbon accounting

Carbon accounting can be defined as the accounting undertaken to measure the amount of GHG (in carbon dioxide equivalents) emitted to or removed from the atmosphere over a specific period of time from applicable activities.

There is an increase in the public disclosures of GHG emissions. Reasons for this include the requirement by regulatory bodies to obtain information related to initiatives such as carbon taxes and emissions trading schemes and for businesses to demonstrate that they are being good corporate citizens. Therefore, the term carbon accounting is often used to describe only the GHG emissions component of the account. Hence, in most cases, it provides a corporate level GHG emission inventory and does not include a carbon mass balance per say.

The DCC has developed frameworks such as the National Carbon Accounting System (NCAS) for estimating and reporting GHG emissions and removals at an enterprise level. The NCAS is a process-based, mass balance, carbon and nitrogen cycling, ecosystem model which has been developed to account for greenhouse gas emissions and removals from land based sectors.

The recognition of climate change as a significant business issue continues to grow. For many Australian organisations the actual process of evaluating the total emissions from operational activities is an important precursor to, and driver for, abatement. Hence, for the purposes of this review, frameworks for accounting of carbon emissions are considered.

GHG emissions at the company or facility level are captured in ISO standards (e.g. ISO 14064: Greenhouse gases Parts 1-3) which provide specifications with guidance for the quantification, monitoring and reporting of GHG emissions and removals.

The Greenhouse Gas Protocol Initiative (GHGP) an international coalition of businesses, non-government organisations, government and inter-governmental organisations convened by the World Business Council for Sustainable Development (WBCSD) and the World Resources Institute (WRI) have developed important tools for standards measurement and reporting of greenhouse gas emissions (WRI 2004). These provide further guidance on measuring and reporting GHG from a facility and company perspective.

The GHGP Initiative aims to develop and promote internationally accepted uniform GHG accounting and reporting standards and/or protocols. It consists of two modules:

- Corporate Accounting and Reporting Standards (Corporate Standard)
- Project Accounting Protocol and Guidelines

The GHGP initiative provides an accounting framework consistent with nearly every GHG standard and program in the world.

The GHG Corporate Module is a tool to provide standards and guidance for companies preparing a GHG inventory: to identify, calculate and report GHG emissions. It is intended to help companies of any size understand their position in relation to the evolving regulatory framework for reducing GHG emissions. It is claimed that the GHGP Corporate Module will improve comparability and enable managers to make informed decisions on carbon risks and opportunities (GHG Protocol Initiative 2004).

Within the GHG Corporate Module, the concept of an operational boundary is used to help companies better manage the full spectrum of risks and opportunities that exist along its value chain (WRI 2004). The operational boundary defines the scope direct and indirect emissions for operations that fall within a company’s established organisational boundary. The protocol recommends that a consistent approach for setting an organisational boundary must be used for accounting and reporting on GHG emissions.

The GHG Protocol differentiates between direct and indirect emissions as follows:

- Direct GHG emissions are from sources that are owned or controlled by the company
- Indirect GHG emissions are a consequence of the activities of the company, but occur at sources owned or controlled by another company (Florence & Ranganathan, 2005).

These are further categorised into three broad scopes:

- Scope 1: all direct GHG emissions
- Scope 2: indirect GHG emissions from consumption of purchased electricity, heat or steam
- Scope 3: other indirect emissions including the extraction and production of materials and fuels, transport related activities in vehicles not owned or controlled by the reporting entity, other electricity activities and outsourced activities

Figure 5 illustrates examples of scope 1, 2 and 3 emissions from business.

The scopes are defined by the International Organisation for Standardisation's Standard for Greenhouse Gases—Part 1: specification with guidance at the organisational level for quantification and reporting of greenhouse gas emissions and removals (ISO 14064-1). Relevant ISO standards are now being adopted in Australia (AS ISO 14064.1-2006, 14064.2-2006, 14064.3-2006). The terms 'scope 1', 'scope 2' and 'scope 3' are well known and used in a number of Australian and international programs and standards.

The GHG Protocol for Project Accounting is a tool for determining the GHG emission reduction benefits of climate change mitigation projects. The development of a consistent approach to GHG project accounting has become increasingly important since the ratification of the Kyoto Protocol (UNFCCC 1997). The Project Protocol includes accounting and reporting standards and guidance for GHG emission reduction projects and land use, land-use change and forestry projects.

The GHG Protocol for Project Accounting was designed by the Greenhouse Gas Protocol Initiative (GHGP) as a tool to be used by project directors and organisations to quantify the GHG emissions from climate change mitigation projects (GHG Projects). It was not intended to be used as a tool to quantify corporate or entity wide GHG reductions (GHG Protocol Initiative 2005).

3.2.1 Emission scope classification

Scope 1 emissions

Scope 1 emissions are direct GHG emissions that occur from sources that are owned or controlled by the enterprise. This does not include direct emissions from the combustion of biomass or other emissions not covered by the Kyoto Protocol. For example, for a grazing property this would include enteric emissions from livestock, GHG emissions from manure, GHG emissions from land use and GHG emissions from usage of fuels (petrol, diesel, etc).

WRI (2004) breaks down Scope 1 emissions into four types. They are:

1. Generation of electricity, heat or steam on site.
2. Physical or chemical processing. This includes waste treatment.

3. Transportation of materials, products, waste or employees. These emissions result from the combustion of fuels in enterprise owned / controlled mobile combustion sources (e.g. trucks, ships, cars).
4. Fugitive emissions. These are intentional or unintentional releases. Examples in the red meat sector could include hydrofluorocarbon (HFC) emissions during the use of refrigeration equipment at abattoirs or methane emissions from manure compost stockpiles.

Fuel used in transport of materials and products occurs off-site and is often done by sub-contractors. There is debate as to where the emissions should be allocated. For example, should the fuel emissions from the transport of cattle from a farm to an abattoir be included in the carbon account of the farm or the abattoir, or neither?

Scope 2 – Electricity indirect GHG emissions

Scope 2 emissions are indirect emissions due to energy usage that is purchased from off-site (primarily electricity, but can also include energy like heating/cooling, or steam) by the enterprise. Scope 2 emissions occur at the facility where the generation of electricity, heating/cooling, or steam takes place. In this case, the emission is caused by the usage of electricity but does not occur on-site. The emission occurs at the electricity generation plant. In Australia, the Scope 2 emissions vary depending on the source of the electricity.

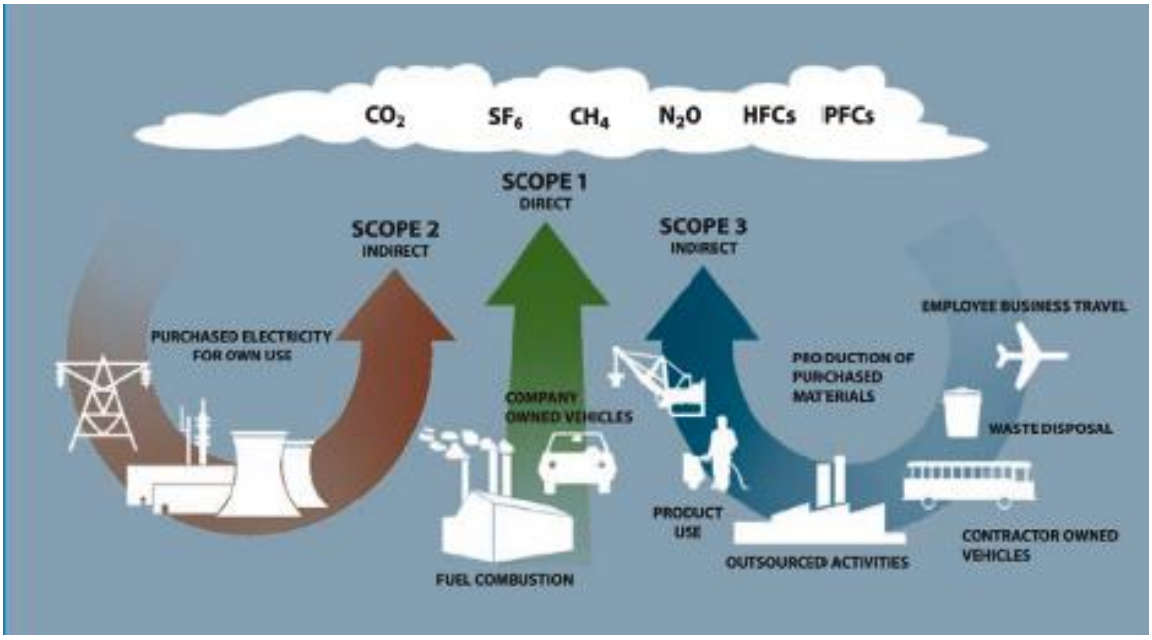

FIGURE 5 – EXAMPLES OF SCOPE 1, SCOPE 2 AND SCOPE 3 EMISSIONS (WRI 2004)

Scope 3 – Other indirect GHG emissions

Scope 3 emissions are other indirect emissions due to the other off-site activities. Scope 3 is much broader and can include anything from employee travel, to "upstream" emissions embedded in products purchased or processed by the enterprise, to “downstream” emissions associated with transporting and disposing of products sold by the enterprise. An example is air travel. Air travel for staff may be an essential component of operating the enterprise but the emissions do not occur on-site. Scope 3 is an optional reporting category but it provides an opportunity for an enterprise to be innovative and inclusive in greenhouse gas management. It can also prevent “*pollution swapping*” and “*green washing*” where a polluting component of an enterprise is out-sourced or hidden to reduce the apparent emission quantum of an enterprise.

A specific Scope 3 issue for agriculture is the “*embodied energy*” and GHG emissions in plant and infrastructure. Embodied energy is the energy used during a product's entire life cycle in order to manufacture, transport, use and dispose of the product (Global Footprint Network 2007). For example, energy is used and GHG emitted in the manufacture of a tractor. This energy is “*embodied energy*” and, arguably, it can be counted as a Scope 3 emission.

3.2.2 National Greenhouse and Energy Reporting System (NGERS)

The *National Greenhouse and Energy Reporting Act 2007* (the NGER Act) establishes a national systematic framework for reporting GHG emissions and makes registration and reporting mandatory for corporations whose energy production, energy use or GHG emissions meet specified thresholds from 1 July 2008. Data reported under the NGER Act will underpin the Australian Government’s proposed Carbon Pollution Reduction Scheme (CPRS) (section 3.2.3). Monitoring, reporting and auditing of businesses' GHG emissions data will be essential to maintain the environmental and financial integrity of the Carbon Pollution Reduction Scheme (DCC 2009a).

The NGERS has two levels of thresholds at which businesses are required to apply for registration and report. These are facility thresholds and corporate thresholds. When a corporation meets a corporate or facility threshold, the corporation must apply for registration and report its GHG emissions and energy data.

The reporting threshold for facilities is 25kt of CO_2-e of GHG emissions or 100TJ of energy. The reporting threshold for corporations in 2008-2009 is 125kt of CO_2-e of GHG emissions or 500TJ of energy. This threshold progressively reduces to 87.5kt of CO_2-e of GHG emissions or 350TJ of energy in 2009-2010 and 50kt of CO_2-e of GHG emissions or 200TJ of energy in 2010-2011.

Direct and, in some cases, indirect GHG emission estimates are required to be reported under the NGER Act. The NGER Act classifies direct and indirect emissions categories in accordance with the international reporting framework prepared by the WRI (2004) and summarised previously in section 3.2.1

Under the NGER Act it is mandatory to report ‘scope 1’ and ‘scope 2’ emissions. However, ‘scope 3’ emissions are not defined under the NGER legislation because it is not mandatory to report them. The NGER initiative directly impacts on corporations and facilities involved in red meat production that are large enough to trip the thresholds.

3.2.3 Carbon Pollution Reduction Scheme (CPRS)

The Australian Government is establishing a CPRS as part of an effective framework for meeting the climate change challenge. The Australian Government is committed to the CPRS and its timeline for the emission trading scheme (ETS) introduction. The NGERS would be the starting framework for monitoring, reporting and assurance under the scheme, and elements of that system would be strengthened to support the scheme (DCC 2008a)

The Australian Government is disposed to include agriculture emissions in the ETS by 2015 and to make a final decision on this in 2013 (DCC 2008a). Even if agricultural businesses are initially excluded from an ETS, they will still likely experience increased input costs such as energy, fuel, labour and fertiliser via those sectors covered by it.

In the advent that agriculture is included in the scheme, red meat production will play an important role with respect to climate change and efforts to address GHG emissions. However, critical research needs to be undertaken that will improve the technical and scientific knowledge about what is happening in biological fluxes. This should not only include environmentally beneficial non-permanent agricultural offset activities such as carbon sequestration through pasture, cropping and soil management but also research into the estimation of emissions from biological fluxes such as breed, genetic manipulation, nutritional management and manure management.

3.3 Carbon footprint

The term “carbon footprint” has gained increased popularity in recent years and is now widely used in government, business and the media. However, the definition of “carbon footprint” is surprisingly vague given the growth in the term’s use in recent years (East 2008).

The term originated from the ecological footprint concept which is still widely used today as a resource management tool. However, in recent years the term carbon footprint has evolved into a concept in its own right (Global Footprint Network 2007).

Carbon footprinting has not been driven by research but rather has been promoted by nongovernmental organisations, companies, and various private initiatives as a tool for the measurement of GHG emissions associated with consumer products (goods and services) (Weidema et al. 2008a) This has resulted in many definitions and suggestions as to how the carbon footprint should be calculated.

East (2008) investigated the definition of ‘carbon footprint’ and found the term had not been adequately defined in scientific literature. Despite the lack of scientific endorsement, the term “carbon footprint” has quickly become a widely accepted “buzz word” to further stimulate consumers’ growing concern for issues related to climate change by describing anything from the narrowest to the widest interpretation of GHG measurement and reduction (East 2008). Therefore, a large range of definitions exist for this term. Some definitions relate to an area of land – hence, the term footprint. For example, one definition says that “*the carbon footprint therefore measures the demand on biocapacity that results from burning fossil fuels in terms of the amount of forest area required to sequester these CO_2 emissions*” (Global Footprint Network 2007). However, most definitions refer to a measure of GHG emissions.

Wiedmann and Minx (2007) suggest that the term “*carbon footprint”* should only be used for analyses that include carbon emissions. The same study showed, however, that most definitions currently include non-carbon emissions and use carbon dioxide (CO_2) equivalent indicators instead.

The UK Carbon Trust define carbon footprint as "*the total set of GHG (greenhouse gas) emissions caused directly and indirectly by an individual, organisation, event or product*" (UK Carbon Trust 2008).

East (2008) provides a review of numerous different definitions of carbon footprint and also provides a definition of carbon footprint to be used in the Australian horticultural sector. The definition provided by East (2008) is:

"A direct measure of greenhouse gas emissions (expressed in tonnes of carbon dioxide [CO_2] equivalents) caused by a defined activity. At a minimum this measurement includes emissions resulting from activities within the control or ownership of the emitter and indirect emissions resulting from the use of purchased electricity"

By this definition, a carbon footprint includes Scope 1 and Scope 2 emissions as a minimum but appears to leave open the opportunity to include Scope 3 emissions. East (2008) notes the lack of precision with this term and suggests that a more rigorous term such as "greenhouse gas accounting" should be used.

In Australia, the weight of evidence suggests that most carbon footprints include Scope 1 and 2 emissions as mandatory, with some including scope 3 emissions with the measurement being expressed in CO_2 equivalents. This ensures that the activity being "footprinted" is consistent with the corporate reporting requirements under the NGERS.

Carbon footprints carry the potential of being a good entry point for increasing consumer awareness and fostering discussions about the environmental impacts of products. However, the most significant issue with the variability in the definition of carbon footprint is that it makes fair comparisons between products impossible if a standard and rigorous definition is not used. In addition, a footprint is by its nature retrospective, i.e. it assesses only what *is* or *was* the size of carbon emissions from a product or company (Grant 2009). In contrast, LCA has a framework for studying proposed systems or system changes through the consequential modelling approach.

Identifying a generally accepted definition of a 'carbon footprint' should consider whether the measurement of a carbon footprint be in tonnes of CO_2 or should it be extended to include a variety of GHG expressed in tonnes of CO_2 equivalents and establishing the boundaries for measuring a carbon footprint is necessary to ensure the accuracy of a footprinting approach. Hence, this raises the issue of whether the measurement of a carbon footprint should include indirect emissions embodied in upstream production processes or only direct emissions within an organisational boundary.

3.4 Life Cycle Assessment (LCA)

The concept of conducting a detailed examination of the life cycle of a product or a process is a relatively recent one which emerged in response to increased environmental awareness on the part of the general public, industry and governments. A number of different terms have been coined to describe the processes involved in conducting this detailed examination. One of the first terms used was *Life Cycle Analysis*, but more recently two terms have come to largely replace that one: *Life Cycle Inventory (LCI)* and *Life Cycle Assessment (LCA).* These better reflect the different stages of the process. Other terms such as *Cradle-to-Grave Analysis*, *Eco-balancing*, and *Material Flow Analysis* are also used.

LCA is a method for analysing processes and models the complex interaction between a product and the environment. It furnishes information on the environmental effects of all the stages of a product's life cycle. This information can be used by governments and by companies as well as by non-government organisations and individual consumers when making decisions related to products. Eco-labelling, product and process improvements, and purchasing decisions, for example, can be supported by LCA.

LCA is a form of cradle-to-grave method of assessing environmental impact. It was developed for use in manufacturing and processing industries and covers the entire life cycle of a product or function, from the extraction and processing of the raw materials needed to make the product to its recycling and disposal.

Because LCA integrates all the environmental impacts produced during the entire life cycle of a product or function, LCA can be used to prevent three common forms of problem shifting:

- From one stage of the life cycle to another.
- From one sort of problem to another.
- From one location to another.

An LCA is an iterative process, in that the assessment is repeated several times, each time in more detail. First, a superficial analysis is made using approximate data; this results in a 'quick-and-dirty' assessment. Although such an analysis is sometimes all that is required, more often this first assessment is used to highlight the points on which to focus to obtain an improved assessment.

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

- **Goal definition and scope**: 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 plant 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.
- **Improvement assessment**: The results are reported in the most informative way possible and the need and opportunities to reduce the impact of the product(s) on the environment are systematically evaluated against the study's goal.

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. 2009a).

An LCA is essentially a quantitative study. Sometimes environmental impacts cannot be quantified due to a lack of data or inadequate impact assessment models. Quantitative analysis requires standardised databases of main processes (energy, transport) and software for managing the study's complexity.

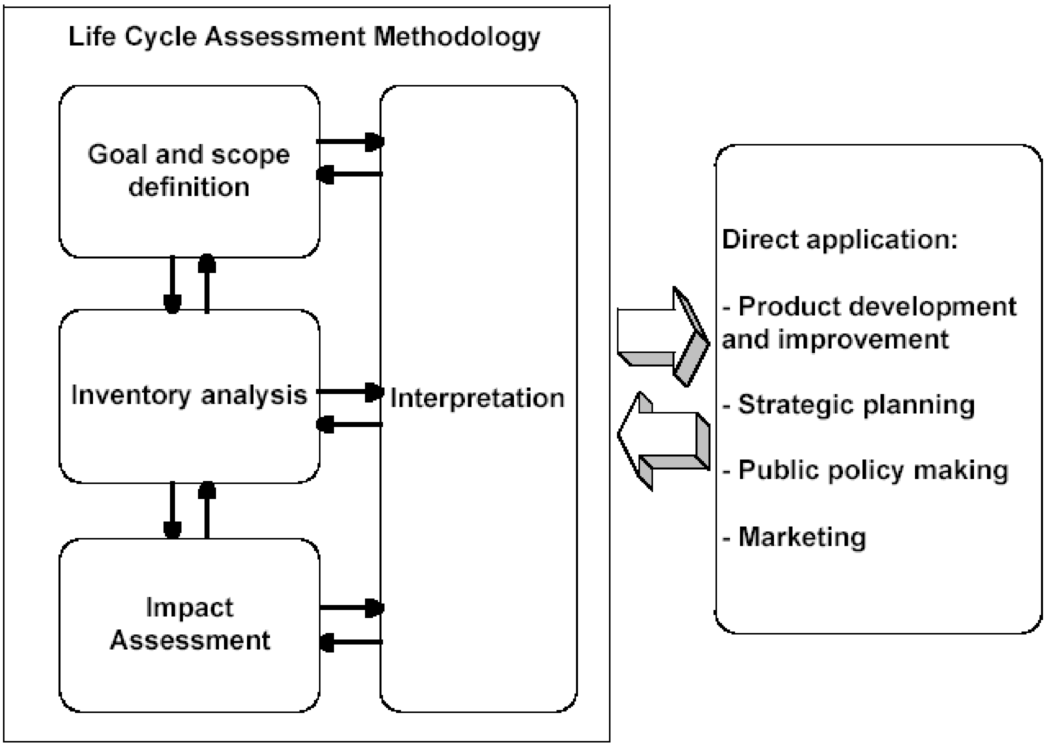

FIGURE 6 – GENERAL FRAMEWORK FOR LCA AND ITS APPLICATION (STANDARDS AUSTRALIA 1998)

Australian rural industries have recognised the importance of LCA studies in agricultural systems and as such are in the process of developing a standardised methodology to assist practitioners undertake LCA studies. This will greatly increase their value by providing results that are comparable between sectors and industries (Harris & Narayanaswamy 2009).

An initial approach to completing a life cycle assessment is a process-based LCA method. In a process-based LCA, one itemises the inputs (materials and energy resources) and the outputs (emissions and wastes to the environment) for a given step in producing a product. Two main issues arise with process-based LCA methods. One is defining the boundary of the analysis. The initial step of a process-based LCA is defining what will be included in the analysis, and what will be excluded and ignored. The other main issue with process-based LCA methods is circularity effects. In our modern world, it takes a lot of the same "stuff" to make other "stuff". For example to make an agricultural machine requires manufacturing equipment. But to make the manufacturing equipment requires other machinery and tools made out of the same product, in this case steel. And to make the steel requires machinery, made out of steel. Effectively, one must have a life cycle inventory of all materials and processes before one can complete a life cycle assessment of any material or process.

3.4.1 Goal and scope definition

The first part of an LCA study consists of defining the goal of the study and its scope. The goal of the study must state the reason for carrying out the study as well as the intended application of the results and the intended audience. The time period that the study encompasses and the geographical region of the agricultural practices under assessment should also be included for agricultural LCAs (Harris & Narayanaswamy 2009).

In the scope of an LCA the following items should be considered and described:

- The function of the product system.
- The functional unit.
- The system boundaries.
- Handling of co-products.
- Type of impact assessment methodology and interpretation to be performed.
- Data requirements.
- Assumptions and limitations.
- Data quality requirements.
- Type of critical review, if any.
- Type and format of the report required for the study.

The scope should describe the depth of the study and show that the purpose can be fulfilled with the actual extent of the limitations. In general, the scope should include water use, energy use and GHG emissions, for the whole life span of the livestock and plants (Harris & Narayanaswamy 2009).

Functional unit

The functional unit is a key element of LCA which has to be clearly defined. The functional unit is a measure of the function of the studied system and it provides a reference to which the inputs and outputs can be related (ISO 14040 2006). This enables comparison of two essential different systems. For example, it would be nonsensical to compare a disposable paper cup with a china cup, given that the life span of the two differs by a factor of at least 100. Instead, the function of the two alternatives, such as drinking one cup of coffee, could be compared. The function to be compared is referred to as the functional unit.

For agricultural products, there are three main types of functional unit that can be used. These include weight (kg product), area (ha) or quality (e.g. protein) based. The choice of functional unit is particularly important when comparing different systems. Harris & Narayanaswamy (2009) provide examples of functional unit choices for rural industries.

The functional unit for the MLA funded LCA projects COMP.094 (Peters et al. 2009a) and FLOT.328 (Davis and Watts 2006) was the delivery of one kilogram of hot standard carcass weight (HSCW) meat at the abattoir. AUS-MEAT is the authority for uniform specifications for meat and livestock in Australia. In March 1987, they introduced the term HSCW as a national standard. The HSCW is the fundamental unit of “over the hooks” selling and is the weight, within two hours of slaughter, of a carcass with standard trim (all fats out). This is a carcass after bleeding, skinning, removal of all internal organs, minimum trimming and removal of head, feet, tail and other items (AUS-MEAT 2001). “Hot” indicates that the meat in question has not entered any chilling operations. In these studies, an output-related functional unit was chosen, rather than an input-related one, in order to describe the human utility of the processes under consideration – the provision of nutrition for people. Although the meat could be served in different ways, this functional unit makes the different processes under consideration “functionally equivalent” from a dietary perspective. It should be noted however that while the functional unit is ‘Hot’ carcass weight, the studies did include the energy required for cooling the carcass.

System boundaries

The system boundaries determine which unit processes to be included in the LCA study. In LCA methodology, usually all inputs and outputs from the system are 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 14040 2006). 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. Harris & Narayanaswamy (2009) suggest that depending on the goal and scope of the study the system boundary should include:

- Pre-farm processes.
- On-farm processes.
- Post farm-gate (processing).
- Post farm-gate (retail).

Harris & Narayanaswamy (2009) have developed a methodology primarily for "cradle-to-farm-gate" studies. Hence, all inputs into on-farm production for each commodity are traced back to primary resources such as coal and crude oil. Their methodology can be easily extended to cradle-to-abattoir or cradle-to-consumer.

Figure 7 shows the generalised system boundary for the red-meat sector as defined for the COMP.094 project (Peters et al. 2009a). Within this boundary, there is a sub-system for the feedlot sector. The boundary chosen here (shown in red on Figure 7) is the feedlot site itself, plus the transport component of bringing cattle and feed into the feedlot and delivering cattle from the feedlot.

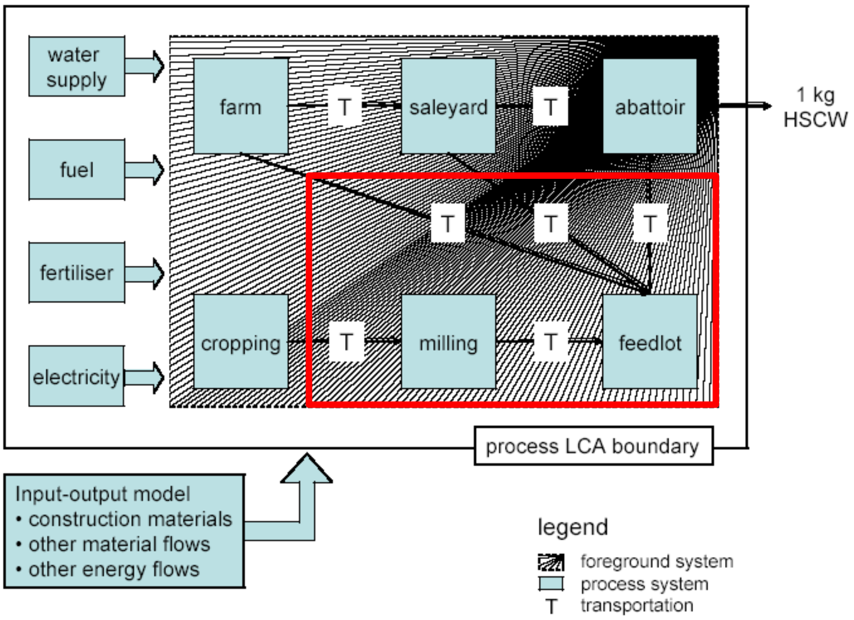

FIGURE 7 – GENERALISED BEEF SUPPLY CHAIN MODEL WITH FEEDLOT SUB-SYSTEM (PETERS ET AL. 2009A)

Data quality requirements

Reliability of the results from LCA studies strongly depends on the extent to which data quality requirements are met. The following parameters should be taken into account:

- Time-related coverage.
- Geographical coverage.
- Technology coverage.
- Precision, completeness and representativeness of the data.
- Consistency and reproducibility of the methods used throughout the data collection.
- Uncertainty of the information and data gaps.

Reusability of data is also highly dependent on sufficient data documentation and is particularly important for comparison between sectors and studies with long time horizons.

3.4.2 Life Cycle Inventory (LCI)

Inventory analysis is the second phase in a life cycle assessment and is concerned with data collection and calculation procedures. LCI comprises all stages dealing with data retrieval and management. The Inventory Analysis phase forms the body of the LCA, as the majority of time and effort in an LCA is spent on Inventory Analysis. As a rule of thumb, 80 % of the time required for an LCA is needed for this phase.

The data collection forms must be properly designed for optimal collection. Subsequently data are validated and related to the functional unit in order to allow the aggregation of results.

The operational steps in preparing a LCI are according to ISO 14041 (Standards Australia 1999):

- Data collection.
- Relating data to unit processes and/or functional unit.
- Data aggregation.
- Refining the system boundaries.

Data issues

For LCA models, as with any other model, it holds that “garbage in = garbage out”. In other words, the parameters of the study and the quality of the data used have a major impact on results, and proper evaluation of data quality is an important step in the LCA.

When considering agricultural products, the majority of impacts (i.e. GWP and water usage) relate to the farm stage of production. For this reason, the detail of data collection on-farm is critical to the results of the LCA. LCA models (and often LCA practitioners) are generally not experts in the fundamental agricultural sciences that drive dominant emissions and resource usage on farms. The importance of this cannot be understated, as many LCA practitioners are used to conducting desk-top analyses with ‘standard’ values drawn from the literature, without a good understanding of the system under study. In some cases, no attempt to collect actual site-based data is made at all. This is clearly not appropriate for agricultural LCA’s, particularly in Australia where the system differs greatly to other countries where examples may be drawn from in the literature.

Many agricultural impacts are derived from highly complex, dynamic systems that are difficult to measure directly. Even production can vary greatly from year to year, and may be complicated by fluctuating livestock numbers and variable growth rates. In most cases, the 'outputs' of emissions must be modelled for the system. Water usage is also regularly modelled, based on such information as 'standard livestock drinking requirements' and 'standard plant evapo-transpiration' rates. The method used to model these data can be the single most critical factor in a study. Consequently, these modelling processes, the data inputs and methods used need to be clearly elaborated in the project methodology. As an example, in previous Australian feedlot LCA research, the BEEF-BAL model was utilised for the feedlot sector of the supply chain to assist in the modelling of waste stream parameters (Davis et al. 2008). This was used to generate manure production and the mass balance of nitrogen, however specific emission factors applied were calculated using the DCC methodology (DCC 2007), which is a 'tier 2' estimation methodology based on the IPCC.

It has been proposed by (Harris and Narayanaswamy 2009) that, wherever possible, real-time data should be collected in conjunction with an LCA. This will not be practical in many cases, as the research required to achieve this is extensive. This said, it must be clearly noted that the results of the whole LCA will only be as good as the accuracy of emissions calculation for the 2-3 major factors (for GWP this is enteric methane, nitrous oxide and possibly manure methane).

Handling of co-products

Allocation processes between primary and co-products can be highly sensitive. Most agricultural systems yield more than one product. For example, dairying produces milk and beef. Materials and energy flows regarding the process as a whole, as well as environmental releases, must be managed in such a way that the appropriate 'environmental burden' is attributed to the different products. The recommended procedure according to ISO-14044 to achieve this is as follows:

- Wherever possible, allocation should be avoided by correct delineation of the system boundary or system expansion)
- Where allocation is not avoidable, inputs and outputs should be partitioned between its different functions or products in a way that reflects the underlying physical relationships between them
- If the latter is not possible, allocation should be carried out based on other existing relationships (e.g. in proportion to the economic value of products)
- The data collection is the most resource consuming part of the LCA. Reuse of data from other studies can simplify the work but this must be made with great care so that the data is representative. The quality aspect is therefore also crucial.

The result of a LCA study involving a multi-input/output system is affected significantly by the choice of the allocation method. For example, for allocation at the point of slaughter results can differ markedly, as shown in Table 4.

TABLE 4 – COMPARISON OF ALLOCATION METHODS AT THE POINT OF SLAUGHTER FOR BEEF CATTLE (GWP)

Liveweight GWP (kg CO_2-e / kg LWT	Mass allocation	Economic allocation (93% to carcass weight) [2]	'Unallocated' – all burden transferred to carcass weight
10	10 x (1/0.779) = 12.8 [1]	10 x 0.93 x (1/0.53) = 17.5	10 x (1/0.53) = 18.9

[1] Variation can occur in the definition of mass allocation. In this instance, we have equally allocated the burdens between all useful by-products (i.e. edible offal, hides, dried blood meal etc). Total yield estimated at 79.5%.

[2] Economic allocation based on 93% of the value ascribed to the carcass, carcass yield = 53% of liveweight).

3.4.3 Impact assessment

Life cycle impact assessment (LCIA) aims to evaluate the magnitude and significance of potential environmental impacts using the results coming out from the LCI phase. The ISO14040 suggests that this phase of an LCA is divided into the following steps:

Mandatory elements:

- Selection of impact categories, category indicators and characterisation models.
- Classification, i.e. assignment of individual inventory parameters to impact categories, e.g. CO_2 is assigned to Global Warming. Common impact categories are Global Warming, Ozone Depletion, Photo-oxidant Formation, Acidification and Eutrophication.
- Characterisation, i.e. conversion of LCI results to common units within each impact category, so that results can be aggregated into category indicator results.

Optional elements:

- Normalisation. The magnitude of the category indicator results is calculated relatively to reference information, e.g. and old products constitutes baseline when assigning a new product.
- Weighting. Indicator results coming from the different impact categories are converted to a common unit by using factors based on value-choices.
- Grouping. The impact categories are assigned into one or more groups sorted after geographic relevance, company priorities etc.

The methodology proposed for rural industries by Harris & Narayanaswamy (2009) focus on water and energy use and GHG emissions.

3.4.4 Interpretation

The aim of the interpretation phase is to reach conclusions and recommendations in accordance with the defined goal and scope of the study. Results from the LCI and LCIA are combined together and reported in order to give a complete and unbiased account of the study. The interpretation is to be made iteratively with the other phases.

The life cycle interpretation of an LCA or an LCI comprises three main elements:

- Identification of the significant issues based on the results of the LCI and LCIA phases of a LCA.
- Evaluation of results, which considers completeness, sensitivity and consistency checks.
- Conclusions and recommendations.

In ISO 14040 standard it is recommended that a critical review should be performed. In addition it is stated that a critical review must have been conducted in order to disclose the results in public.

3.5 Comparison of GHG methodologies

The increasing awareness about environmental impacts, especially climate change, has led to many initiatives to try to mitigate GHG emissions. Examples include international agreements such as the United Nations Kyoto Protocol and emission trading schemes (carbon trading). These activities require the ability to accurately measure GHG emissions and sinks.

There are a number of methodologies which provide a framework for the estimation of GHG emissions. The most appropriate assessment methodology for the red meat industry will depend on what decisions and above all, whose decisions the information is intended to support.

At the industry level, emissions are grouped by the NGGI *by emission source*, leading to a ‘sector-by-sector’ and emission-by-emission view of the nation and the red meat industries. Reporting does not take into account production efficiency, though the emission profile may change if performance is deemed to have improved across the whole industry for a given emission. It will not be obvious from this approach to emission estimation and reporting whether changes are the result of improved performance or simply lower emissions because of, for example, reduced numbers of livestock in the national inventory.

Business GHG accounting and reporting practices will be an important part of the red meat industry because they are regulated by legislation (i.e. the NGERS) and are likely to form the basis for ongoing reporting and emission obligations through the carbon pollution reduction scheme (CPRS). For this reason, economic modelling is more likely to investigate the impact at the business level, and businesses will adapt to regulations through a variety of approaches. The general business framework is comprehensive when all ‘scopes’ are considered; though in practice this is rarely done at the business level.

Carbon footprinting has arisen to provide a tool for the measurement of GHG emissions associated with consumer products and to assign these products with a carbon or environmental label. Development has not been driven by research but has rather been promoted by nongovernmental organisations, companies, and various private initiatives, resulting in many definitions of the term, and a variety of methods for accounting.

Carbon footprinting is in some respects an intermediate between business accounting and life cycle assessment, though it generally suffers the weaknesses inherent with trying to hybridise two existing frameworks. It is not considered to be as thorough or robust as LCA at the product level. Grant (2009) provides a comprehensive comparison of the differences in structure, method and results between carbon footprints and LCA (outlined in Table 5).

TABLE 5 – COMPARATIVE DIFFERENCES BETWEEN CARBON FOOTPRINT AND LCA (GRANT 2009)

Item	Carbon Footprinting	Life Cycle Assessment
Structure		
Purpose	To quantify carbon emissions from the production of a product or service, or from an organisation.	To determine the potential environmental impact of a product or system from cradle to grave.
Standardisation	Evolving and possibly competing standards are currently being developed over a short timeframe.	Standards have developed over a 10 year period, leading to a consensus position particularly about abuse of the tool for comparative assertions.
Application of standards	Relatively poor at the early stage as standards still lack maturity	Improving use of standards in formal practice. Informal practice still often breaches the standards
Regulation of practice	Regulated through government schemes such as 'Greenhouse Friendly' and soon to be regulated through government scheme.	Not regulated. Standards are mostly voluntary. Environmental product declarations recently legislated for all products in France.
Method		
Scope	Practice varies between onsite emission and electricity (scope 1&2) and inclusion of offsite inputs (scope 3 emissions). Background infrastructure and service input are not routinely included.	All major material and energy inputs are included. Newer databases routinely include capital and infrastructure.
Calculated against	Product, service or organisation or some mix of these.	Calculated against the functional unit.
Modelling approach	No consistent approach, although some practical consensus is being developed. Carbon offsets are based on "additionality" (consequential modelling). PAS 2050 uses ISO LCA standards approach.	Hierarchy and method for dealing with co-production. Consequential and attributional methods (marginal and average) used in LCA
Timeframe	Timing of emission releases is sometimes important. By default 100 years is used for calculation of warming factors.	Timeframe is normally long, from 100 to 500 years with some impact methods calculated over thousands of years.
Indicators	Greenhouse gas emissions	Often based on multiple impacts, although evaluation of greenhouse gas impacts alone is common. Impact categories should be those related to the product system under study.
Results		
Interpretation	The greenhouse result is the main focus which can then best be offset or tracked over time to look for reductions.	Results are interpreted though formal procedure to identify underlying causes and verify the data driving the main results.
Comparative	Not usually comparative, but may be done when Product Category Rules (PCR) are used.	Mostly comparative assessment, either between products or alternate production approaches to a single product.

4 Review of LCA literature

4.1 Australian agricultural LCA research

Most major agricultural industries in Australia have completed, or are in the process of conducting LCAs. Completed studies in the last 10 years include the dairy industry (Lundie et al. 2003), red meat (Peters et al. 2009a) grains (wheat, barley, canola - Narayanaswamy et al. 2004, maize - Beer et al. 2005) and pork (Wiedemann et al. 2010b).

Additionally, Ridoutt et al. (2009a, 2009b) have published work on water footprinting for the Mars group in Australia covering several agri-food supply chains. Other private work has been carried out for some industries but these are not available in the public literature.

Industries currently undertaking LCA research include chicken meat, eggs, cotton and extensive beef.

Due to the rapid development of LCA methodology and knowledge growth in this area, older research should be subjected to a greater level of scrutiny when comparing results.

4.2 International beef LCA research

The environmental impact of producing beef for human consumption has been investigated by various life cycle assessments, material flows, carbon footprint and food mile studies. Most studies indicate that the production phase of beef contributes the largest proportion of environmental impact of the end product (i.e. Peters et al. 2009a). Studies cover different production systems including feedlot and pastoral production, organic and conventional farming practices, and extensive versus intensive systems. Only one study was found that conducted a gate to gate assessment, so these studies are difficult to compare with this study. However, they do provide valuable context.

Studies that report individual emission sources only, have not been covered here. Some studies have investigated beef co-products such as leather for footwear (Mila I Canals et al. 2002 and Milà I Canals et al. 1998), while others investigate the GHG emissions of beef as part of a meal (i.e. Sonnesson et al. 2005).

Most studies reviewed report impacts for GHG emissions (using the impact category GWP) and energy. No studies covered water usage, hence water will be covered separately in this literature review.

The studies in the literature have been reviewed with respect to the assessment method and system boundary used, the production system investigated and the country of origin (which will influence the emission factors for some key parameters such as N_2O).

LCA methods and system boundaries

In general, studies reviewed followed an LCA methodology based on ISO and IPCC standards. In order to improve specific elements of the study at the farm level, Casey and Holden (2006) used a nutrition software package (RUMNUT) to estimate enteric methane emissions and Peters et al. (2009a) used the mass balance program BEEFBAL (McGahan et al. 2004) to model livestock performance and nutrient flows in the feedlot sector of the supply chain. Nemry et al. (2001) used a materials flow approach to calculate GHG emissions using the CORELLI model.

The system boundaries for the majority of studies are from cradle to farm-gate (do not include meat processing). There are some exceptions, Goldberg (2008) which expands the study by

Barber et al. (2007), includes transport to the meat processer and processing, and transport from New Zealand to a London port). Nemry et al. (2001) is from cradle to retailer and Weidema et al. (2008b) is from cradle to grave.

Ogino et al. (2004) was the only study found investigating a feedlot system in a gate to gate study, beginning with calves at 8 months of age through to slaughter. This study does not include the embodied emissions from the production of the calves. Peters et al. (2009a) conducted a retrospective study using a ‘cradle-to-processor’ supply chain for beef and lamb production in 2 reference years, 2002 and 2004. At the farm level, this study used the physical farm boundary as a system boundary, which in some cases meant the inclusion of alternative agricultural products such as wheat (in the WA supply chain) and sheep / wool (NSW supply chain) which required an additional allocation step to apportion burdens between multiple products. For the Victorian supply chain in this study, beef production in one year (2002) represented the production of young cattle from 7 months to 20 months (embodied emissions from the production of calves not included) while in the second study year (2004), this supply chain had moved to producing calves from breeding through to finishing which resulted in 43% higher emissions per kilogram of beef produced (Peters et al. 2009a).

From the studies that incorporated meat processing, this stage contributed from 1% (Goldberg 2008) to 8.5 % (Peters et al. 2009a) of the total GHG emissions to the boundary of the processer. The greater influence of the meat processing stage will be the result of the allocation process applied to the breakdown of the animal at the point of slaughter.

Management of co-products

Co-products in LCA are handled in a number of ways (discussed previously in this report). Depending on the method used, considerable differences in the final result can be achieved. For example, co-products at the point of slaughter (meat, offal, hides etc) have been dealt with in the following ways for beef:

- System expansion – Weidema et al. (2008b) handled co-products at the point of slaughter by expanding the system to include the *avoided emissions* from a similar product that could be substituted for the relevant by-products
- Mass allocation – Peters et al. (2009a) handled co-products at the point of slaughter using a mass allocation approach, where environmental burdens are attributed to all products based on the mass of the product. The problem with this approach is that it will apply environmental burdens to what may be considered ‘waste’ products, and very low value products such as blood and bone meal.
- Williams et al. (2006) applied an economic allocation process.

Another allocation issue has been raised in several studies that investigate beef production from dairy herds. Incorporating source calves from the dairy industry has been found to reduce the GHG emissions intensity of beef as their emissions are partly allocated to milk production. Williams et al. (2006) applied an economic allocation to milk and beef for their system. Cederberg & Stadig (2003) allocated 19% of their beef production to milk production by using dairy calves. Vergé et al (2008) estimated that replacing one-fifth of beef calves in Canada with dairy calves would reduce their beef GHG emission intensity by 10%.

Production systems

Three studies compared organic and non-organic production systems. Casey and Holden (2006) reported that the organic system had lower GHG emissions per kg LWT and per hectare of land used, whereas the Australian study (Peters et al. 2009a) and the UK/Wales study (Williams et al. 2006) reported higher GHG emissions for organic production systems. The UK/Wales organic production systems also had higher land use, acidification and nitrogen losses. Poorer results from organic systems are typically related to the lower productivity of these systems, which will result in higher enteric methane emissions per kilogram of beef produced.

Four studies reported on production systems that were pasture based and did not include grain feeding. These studies report a wide range in GHG emissions from 8.4 kg CO_2-e / kg carcass weight (CW) (Sahelian), 20.5 kg CO_2-e / kg CW (AUS), 22.2 CO_2-e / kg CW (EU) and 28 kg CO_2-e / kg CW (Brazil). The Sahelian case study only included emissions from enteric losses and periodic grassland burning (Subak 1999) whereas the EU scenario (Cederberg & Stadig 2003) was based on diet of high quality pasture and silage and included emissions from enteric fermentation, manure management and replacement heifer production indicating a considerably more comprehensive study. The Brazilian study (Cederberg et al. 2009) covered all livestock and energy related emissions through to the farm gate, and extended the supply chain through to the delivery of boxed beef to Europe. This study will also be extended to incorporate the impact of land use change (deforestation to expand pasture land for beef production) on overall GHG emissions; however the results of this part of the study are not yet available.

Studies that incorporated intensive production (grain feeding) include Ogino et al. (2004), Weidema et al. (2008b), Vergé et al. (2008) and Peters et al. (2009a). Of these, Vergé et al. (2008) presented the lowest emissions followed by Peters et al. (2009a). Peters et al. (2009a) indicated that finishing beef on grain as preferred to pasture resulted in the lower emissions for an Australian supply chain.

The level of detail provided in the literature on the production system studied and assumptions used varied greatly. Some studies did not specify what production systems were used at all (e.g. Barber et al. 2007; Nemry et al. 2001), while others gave a high level of detail. Considering the large differences that can exist between agricultural systems with and between nations, the rigour of on-farm data collection is highly relevant. Data collection methods applied by some LCA researchers rely heavily on desktop analysis and economic input-output data. Considering the dominance of the on-farm emissions (particularly enteric methane and nitrous oxide) the appropriateness of this approach is questionable. To date, few studies have incorporated alternative scenarios based on ‘best practice’ management for the reduction of major GHG on-farm, which would provide a valuable insight into the impact of such practices.

In all studies that broke down the GHG emissions into separate sources (CH_4, N_2O and CO_2), methane was the largest contributor, followed by nitrous oxide emissions (see Table 6). This is likely to be the same for all other case studies, which did not detail the contributions to CH_4 emissions. Increasing the digestibility of the diet through grain feeding was found to reduce methane emissions in some studies, however the larger proportion of crops required to achieve this may affect other sustainability issues. Generally the emissions associated with the land use change resulting from increased demand for products such as soybean is not accounted for, though this may be quite a significant source of GHG emissions (Garnett 2008).

Comparison between countries

To compare the GHG emissions from different countries we must keep several factors in perspective that may alter the results. This includes the age of slaughter, the source of feed, housing requirements, breed, feed efficiency, manure management and land-use requirements.

The Japanese Wagyu feedlot beef has the highest GHG emissions (Ogino et al. 2004). However, this is more strongly related to the sources of feed (imported from overseas) and the expected lower growth rates for cattle fed from 8 months through to 30 months. In comparison, the Australian feedlot system feeds export steers for only 4 months (Peters et al. 2009a). Decreasing the feeding length in the Japanese system by one month was found to reduce GHG emissions by 4.1 % (Ogino et al. 2004) and altering the source of feed ingredients from imports from the USA to local sources was also found to reduce GHG emissions (Kaku et al. 2006).

European production systems generally have housing requirements during winter whereas Australia, New Zealand and Africa do not. Eutrophication is a very large European issue, whereas in Australia water use efficiency is of far greater importance. Some countries have a higher proportion of dairy cows to suckler-beef, which will reduce the GHG emissions allocated to beef production if dairy culls are included in the system.

Some studies also presented data for primary energy use for beef production (see Table 7). These findings make an interesting comparison between countries and show the considerable differences between management practices across the global beef industry. In general, Australian and NZ production from pasture leads to lower primary energy usage than most European studies (Table 7). Energy usage was higher when Australian cattle were fed through a feedlot (Peters et al. 2009a) but this was still lower than several results from overseas. Energy, while only a minor contributor to GHG, is also a resource usage impact in its own right, particularly considering the limited supply of fossil fuel worldwide.

When compared to a relatively similar grass fed rangeland system (Brazil), overall GHG emissions were lower for Australian organic production (20.5 kg CO_2-e / kg CWT – Peters et al. (2009a) compared to 28 kg CO_2-e / kg CWT – Cederberg et al. (2009)). Cederberg et al. (2009) identified enteric methane as the largest source of GHG emissions (76%) with nitrous oxide from pastures contributing 22%. As with the Australian study, GHG from energy usage contributed a relatively small proportion of overall emissions.

Completeness

The quality of data and the extent of inventory of each study are highly variable. Several studies included the embodied energy and emissions from the production of farm machinery and/or buildings (Barber et al. 2007; Williams et al. 2006; Vergé et al. 2008). Other studies such as Peters et al. (2009) and Weidema et al. (2008b) used economic input-output data to account for products and services that are difficult to quantify using standard inventory and modelling practices. In the case of Weidema et al. (2008b) this resulted in considerably more emissions and particularly energy used (see Table 7). Subak (1999) was the only study to include CO_2-e emissions from carbon offset opportunities that were forgone by using land for feed production. The Belgium inventory (Nemry et al. 2001) includes emissions for breeding (which are quite substantial) but not specifically for animal production (i.e. enteric methane, manure management etc). The USA study (Subak 1999) uses diet composition and weight gain data from 1987 and 1989, which is likely to be outdated, as most feedlots have improved their feed efficiency since this time, which is likely to result in reduced methane emissions. The Canadian study, which is assumed to have a very similar production system to the USA, found a decrease in GHG emission intensities from 16.4 kg CO_2e/kg LWT in 1981 to 10.4 kg CO_2e in 2001. The decrease was attributed to reduced fossil fuel use from the adoption of low tillage practices for feed production and a shift towards low roughage, higher energy intensity rations that reduced methane production (Vergé et al. 2008).

TABLE 6 – GLOBAL WARMING POTENTIAL OF BEEF PRODUCED FROM DIFFERENT COUNTRIES AND PRODUCTION SYSTEMS ASSESSED USING LCA

Reference	Country	System	GHG	Results on standard basis kg CO_2e/kg HSCW – unallocated[1]
Casey and Holden (2006)	Ireland	Conventional	**total**	**24.0**
		Rural EPS	**total**	**22.6**
		Organic	**total**	**20.6**
Cederberg & Stadig (2003)	EU	Organic/pasture	**total**	**17.2**
Goldberg (2008)	New Zealand	Conventional	**total**	**8.8**
Nemry et al. (2001)	Belgium	Not reported	CH_4	6.4
			N_2O	5.1
			CO_2	3.4
			total	**14.8**
Peters et al. (2009a)	AUS (VIC)	Organic (2004)	**total**	**18.1**
	AUS (NSW 2002)	Pasture/feedlot	**total**	**15**
	AUS (NSW 2004)	Pasture/feedlot	**total**	**15.4**
Subak (1999)	USA	Pasture/feedlot	**total**	**14.8**
	Sahelian	Pasture	**total**	**8.4**
Vergé et al. (2008)	Canada	Pasture/feedlot – from national statistics	CH_4	10.3
			N_2O	6.7
			CO_2	1.9
			total	**18.9**
Williams et al. (2006)	UK	Mixed sourcing of beef and dairy calves (conventional production)	**total**	**17.0**
		Single enterprise beef production	**Total**	**27.2**
Weidema et al. (2008b)	EU-27	Feedlot/pasture	**total**	**28.7**
Cederberg et al. (2009)	Brazil	Pasture	**total**	**28.0**
Ogino et al. (2004)	Japan	Long-fed feedlot – gate to gate	**total – feedlot only**	**32.3**

[1] For comparison between studies, data have been re-analysed to attribute all of the environmental burden to carcass weight at the point of slaughter. In reality there are several valuable by-products (i.e. hides, edible offal), however for the sake of comparison between studies this is a useful approach. In several studies the allocation processes used were not clear, but wherever possible results were checked by re-analysing primary data. Where data could not be re-allocated the results were assumed to be on an unallocated basis. The reader is directed to the original references for further information.

TABLE 7 – PRIMARY ENERGY USE OF BEEF PRODUCED FROM DIFFERENT COUNTRIES AND PRODUCTION SYSTEMS ASSESSED USING LCA

Reference	Country	Production system	Energy Use (MJ / kg CW)
Peters et al. (2009a)	AUS (VIC 2004)	Organic	20.2
Peters et al. (2009a)	AUS (NSW 2002)	Pasture/feedlot	24.4
Peters et al. (2009a)	AUS (NSW 2004)	Pasture/feedlot	20.0
Barber et al. (2007)	NZ	Pasture	11.9
Australian and NZ average			**19.1**
Cederberg and Stadig (2003)	Sweden	Not known	78.1
Williams et al. (2006)	UK/Wales	Mixed sourcing of beef and dairy calves (conventional production)	29.9
Williams et al. (2006)	UK/Wales	Single enterprise beef production	40.7
Average of 3 overseas studies			**49.6**

4.3 Australian feedlot industry research

Few studies to date have conducted a comprehensive inventory of GHG emissions from feedlots. One project, FLOT.328 (Davis & Watts 2006) completed an inventory of the emissions occurring at the feedlot site alone as a result of direct activities (i.e. enteric methane emissions, manure emissions, energy emissions) but did not report these in an LCA context (i.e. including 'upstream' emissions, particularly from grain used in the feedlot ration). Never-the-less, Davis & Watts (2006) reported the following breakdown of emission sources (Figure 8).

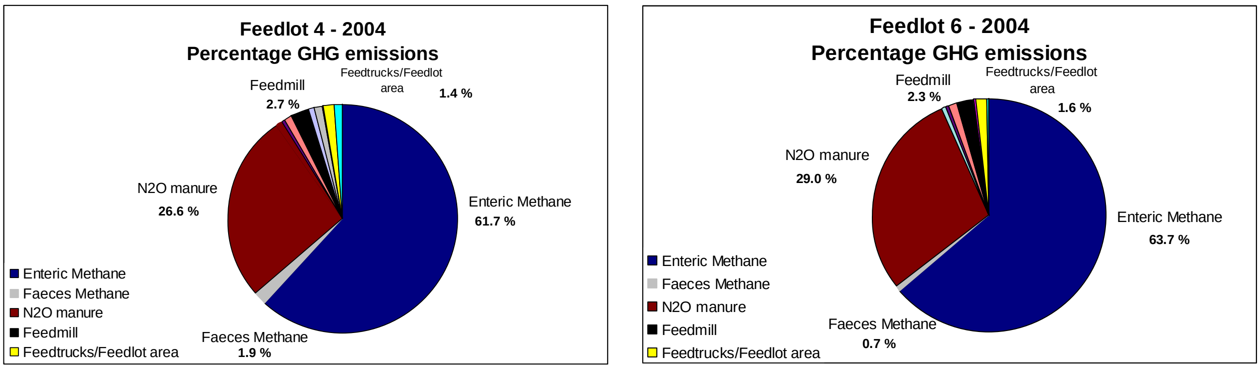

FIGURE 8 – SOURCES OF GHG EMISSIONS FOR TWO AUSTRALIAN FEEDLOTS (DAVIS & WATTS 2006)

From Figure 8 it can be seen that enteric and manure nitrous oxide emissions dominate overall emissions. However, when the upstream emissions from grain are included these proportions will be substantially altered.

A purpose of this project is also to differentiate between flows of GHG from the manure management system to improve prospective research in this area. Hence, the majority of the attention in this scoping study is directed to these emission sources rather than enteric methane or energy related emissions. It is noted that, as the largest single emission source from beef cattle production, enteric methane has received the bulk of research to date into emission estimation and mitigation strategies.

This work has not been reviewed as part of the current project in lieu of the extensive review recently conducted for MLA by Cottle & Nolan (2009). Selected data and conclusions from this review have been used where relevant in this project and are referenced accordingly.

The enteric methane emission formulas used in this project follow the DCC (2007a) methodology and are detailed in section 7.1.

4.3.1 Manure emissions

Manure emissions, particularly nitrous oxide were reported to make up a significant proportion of overall feedlot emissions in prior research (Davis & Watts 2006). However, this project relied on modelling to determine emissions and was not validated by on-ground research. The literature review by Redding (2010) found some data on specific emissions, but no studies where a the whole nitrogen and methane flow pathways and fates were accounted for. Hence, no holistic mass balance approach to estimating feedlot emissions has been presented in the literature, despite this being the recommended approach from the IPCC (Dong et al. 2006).

For this project, two approaches were used to quantify individual emission sources, i) the DCC methodology (DCC 2007a) and ii) a theoretical mass balance approach, which follows the flows of nitrogen and carbon right through the feedlot system, using emission factors taken from the DCC (where available) and supplemented with IPCC and literature sources. Literature used to compile the theoretical mass balance are tabulated in the methodology section of this report.

4.3.2 Australian feedlot water usage research

Water assessment for livestock is the subject of extensive debate in the media and in the scientific literature. This has led to a considerable range in values for ‘water usage’ for beef cattle. Australian research (Davis & Watts 2006; Davis et al. 2008; Peters et al. 2009a, b) have generated conservative results when compared to the global literature, primarily because the focus of these research projects has been on engineered water (taken from dams, rivers, bores etc) and rainwater has not been included within the assessments. This represents a partial ‘blue water’ assessment (discussed in the following section).

Davis & Watts (2006) report water use from 9 Australian feedlots ranging from 34 L / kg HSCW gain to 381 L / kg HSCW gain, with a median value of 73 L / HSCW gain. The main influence on total water use was the quantity of water used to dilute effluent irrigation water (corresponding to the highest value reported), followed by drinking water.

Davis et al. (2008) investigated water use at seven feedlots, reporting that total annual clean water use (without dilution of effluent) over two years ranged from 49.5 L/kg HSCW gain to 51.5 L/kg HSCW gain. Of this, some 90% is used for drinking water, with minor uses for feed milling and cleaning around the feedlot.

4.3.3 Energy research in the feedlot industry

Energy usage in the feedlot industry has been the subject of two MLA research projects, FLOT.328 (Davis & Watts 2006) and B.FLT.0339 (Davis et al. 2008).

Davis & Watts (2006) surveyed 9 Australian feedlots, collecting both indirect and direct energy usage data. Indirect energy use included that consumed in the transportation of cattle and commodities, while direct energy usage data was associated with feed processing, feed delivery, water supply, irrigation, administration and other farming activities. The survey covered two production years (2002 and 2004). Total energy consumption ranged from 1.14 to 17.8 MJ/kg HSCW gain in 2002, and from 1.4 to 12.8 MJ/kg HSCW gain in 2004.

Feed processing was found to be the single largest consumer of energy in the feedlot sub-system, accounting for up to 70% of the total energy consumption. Energy consumption ranged from 0.25 MJ/kg HSCW gain (tempering) to 4.4 MJ/kg HSCW gain (steam flaking).

Davis et al. (2008) extended energy usage research for a further two years in the seven of the same feedlots surveyed by Davis & Watts (2006). Energy usage was assessed with a detailed monitoring and recording program, incorporating metering of all major energy uses at the feedlot and monthly recording. The total annual energy usage in 2007-2008 ranged from 18.5 MJ/kg HSCW gain to 82.9 MJ/kg HSCW gain.

These projects focussed on direct energy usage at the feedlot site together with selected transport data (for cattle and commodities). However, other upstream energy usage (such as the energy required to supply diesel to the feedlot via extraction and transport etc) were not included. For this reason, results from these studies are only a subset of the energy usage assessed in a full LCA and cannot be directly compared.

5 Water methodology research

Methodology development for LCA in Australian agriculture was enhanced by the funding of a LCA methodology project by the RIRDC (Harris and Narayanaswamy 2009). This project focussed on GHG, energy and water assessment. In general this document represents a honing of the ISO standards for LCA (ISO 14040-14044) with some specification with regards to on-farm data collection and the handling of water. The methodology for handling water is summarised in the following section together with a broader literature review conducted by Wiedemann et al. (2010a).

Within the field of LCA, water methodology is expanding rapidly both in Australia and internationally, though to date an agreed direction has not been established. Methodology for the assessment of water was progressed by a project commissioned by RIRDC on behalf of a number of Australian agricultural industries, including MLA. However, this project (Harris & Narayanaswamy 2009) did not achieve a consensus on water usage methodology, partly because of the rapid methodological development in this area. To date the proposed methodology has not been adopted by Australian LCA projects and is not considered state of the art (S. Winter, pers. comm.).

A review of the range of methods for assessing water use and the current state-of-the-art in the field of LCA was undertaken for MLA in the project B.CCH.2022. This report (Wiedemann et al. 2010a) is comprehensive in its explanation of the multiple methods for water assessment and explains the variability in water usage data presented in the literature. The methodology section from this report is summarised here to enable a better understanding of the terminology and approach used for water usage in this project compared to the literature.

Water usage is calculated using a variety of definitions and methods, resulting in highly variable results for water use in beef production in the literature. These approaches can be grouped under two general areas, **water engineering** (water balance principles) or **virtual water / water footprint** methodologies. More recently, methodology development in LCA has advanced, largely as an integration of the above two approaches.

5.1 Water engineering

Water engineering is the traditional approach to water use assessment adopted by private enterprises and governments to define the quantity of water used in a particular locality (i.e. a farm, catchment or state). In Australia the Bureau of Statistics (ABS) provides definitions for the consideration of water use, and engineers apply water balances (or partial water balances) to determine water use within a given system.

ABS definitions of water usage

The ABS defines water use as *the sum of distributed water use, self-extracted water use and reuse water use.* This is compatible with data available to most water users (i.e. water bills for reticulated supply, meter readings for bores).

“Distributed” and “self-extracted” water uses are defined as water supplied from engineered delivery systems. Delivery systems vary greatly in size and degree of infrastructure, incorporating a range of systems, from sub-artesian groundwater extraction to water supply from rivers or state-owned dams.

Water is classified as “distributed” if the water is purchased, or “self-extracted”. For water to be considered “used”, it has either been transferred from its natural watercourse or extracted from groundwater. Hence, water from small overland flow dams may not be considered in water use estimations.

“Reuse water” refers to any drainage, waste or storm water that has been used more than once without being first discharged to the environment. It can refer to both treated and untreated water.

Delineation is also made between the terms *consumption* and *use*. Water consumption differs from water use in the sense that it represents the net water balance for an activity *less* the amount of water passed on for other uses. For example; a hydroelectric power station has a high water use - accounting for all of the water which enters the facility - but a very low water consumption, since almost all of the water ‘used’ is discharged downstream for other uses.

The ABS definition of water use includes the volume of water lost through supply systems. The attribution of this loss volume to suppliers and consumers depends on the origin of the loss. For example, distribution system losses are considered to be a form of use by the *supplier* and metering losses are considered to be a form of use by the *consumer*.

This definition was used by the first MLA LCA project (Peters et al. 2009a), and was the basis for the feedlot water usage research by Davis & Watts (2006).

Water balances

A water balance is a method of accounting for all water in a system by measuring or estimating water inputs and outputs. In its simplest sense, water use is defined as the sum of the water outputs from a system, or the sum of the water inputs minus water captured in storage within the system.

Within the definition of water use, delineation can be made between *beneficial* uses of water and *non-beneficial* uses, or losses. Water ‘use’ may include both beneficial and non-beneficial uses depending on the purpose of the balance calculations.

The strength of this approach is that it provides a full assessment of water movements attributable to a system, identifying where improvements can be made by reducing or eliminating losses. Water balances can be applied at any scale depending on the resolution of input data and the required resolution of output data.

If water use is to be attributed to a product (a kilogram of grain or beef) the general approach would be to account for all ‘system’ water inputs (from watercourses, storages, groundwater etc) which are directly related to production. Rainwater may also be included in the balance, though this is identified separately. Where rainfall is captured on a site because of environmental considerations, this water is considered a water ‘use’ and is attributed to the product, because the water is being restricted from other uses in the environment. A water balance for open systems (such as a farm) will generally include rainwater for completeness, however this is not reported in the balance as a contributor to water use. For example, when water use is quoted for cotton, this generally represents the volume of water that was irrigated onto the field to grow the crop, not the water actually used by the plants (which would include rainfall that fell during the growing season and stored soil moisture that was present prior to planting).

Davis et al. (2008) used a water balance approach in their assessment of water use at 7 Australian feedlots. However, this did not include rainfall and did not identify loss pathways.

5.2 Virtual water and water footprinting

The Virtual Water (VW) concept was first proposed by Allan (1998) to describe the water required to produce tradable commodities (particularly food) in water stressed economies. Hoekstra (2003) has identified two definitions of VW, i) the volume of water that was required to produce a product *in reality* (i.e. for wheat produced in Australia and exported to the middle east, the VW by this definition is *the water required to produce the crop in Australia* in the year of production), or ii) the volume of water that *would have been required* to produce the product in the country of interest (i.e. for the above example, this would represent the volume of water that *would have been required to produce the same amount of wheat in the Middle East* where the wheat is imported to). The lack of consensus in definitions for VW contributes to variable figures within the literature depending on the approach adopted.

The definition of VW has been expanded to differentiate between water depending on source and transferability. Falkenmark describes water in 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'. This distinction between blue and green water is very useful when considering water resources and water scarcity, and offers a clear way to interpret the variance in 'water usage' figures presented in the literature for meat production. Generally 'blue' water usage (as calculated using a water balance or from metered data) for meat production is quite low (10-50 L / kilogram of meat such as beef or chicken), while estimates including 'green' water may be several thousand litres per kilogram of meat, because of the rainwater used when growing the feed (pasture and/or grain).

Virtual water estimates are generally made retrospectively, based on the water requirements (evapotranspiration) of crop production and animal requirements in specific regions. As noted, this may represent an estimate of the water actually required to grow the given product, or an estimate of the avoided water, the water that would have been required to grow the crop in the country of interest. Methodologies for the calculation of virtual water using both approaches have been reported by Hoekstra (Hoekstra & Hung 2002, 2005, Chapagain & Hoekstra (2003) and Renault (Renault 2003; Zimmer & Renault 2003).

Hoekstra (Hoekstra & Hung 2002) introduced the term 'water footprint' to refine their assessments of virtual water. These authors present their data interchangeably under the headings 'virtual water' and 'water footprint'. However, the 'water footprint' term is a useful distinction for describing the methodology presented by these authors, and relates to the virtual water use of a specific product from a specific country. The virtual water use/water footprint of a range of agricultural products has been compiled by Hoekstra and Chapagain (2007). Results from these authors are presented in Table 8.

TABLE 8 – VIRTUAL WATER USE ESTIMATES FOR ALTERNATIVE PROTEIN SOURCES

Species	L / kg (Australian estimates)	L / kg (World average)
Beef	17,112	15,497
Chicken meat	2,914	3,918
Pork	5,909	4,856
Sheep meat	6,947	6,143
Soybeans	2,106	1,789

Source: Hoekstra & Chapagain (2007).

Interestingly, soybeans are not significantly superior in terms of water use to the more efficient meat products, particularly if the protein content were taken into account.

The virtual water and water footprint tools may be useful for minimising local water impacts through the trade of food commodities between regions with differing levels of water scarcity, however these concepts are misleading when used to comment on the resource usage or environmental impacts of a production system. For example, a hydro-electric power plant may 'use' more water than a coal burning power plant, but the impact of this water usage may be quite different. For example, the hydro-electricity plant may not reduce the quantity or quality of water from other users downstream, while the coal burning power plant may evaporate water, thereby removing it from the immediate water stream.

5.3 LCA water usage and impact categories

Water usage in agricultural LCA is the focus of on-going debate over methodology both in Australia and internationally. To date, there is no established methodology or suite of methodologies that have been established for agricultural LCA, though several have been proposed in Australia (Harris & Narayanaswamy 2009) and internationally (Owens 2002; Mila i Canals et al. 2009; Pfister et al. 2009; Ridoutt et al. 2009).

Life cycle assessment has not, as a rule, included water use within its framework of assessment. Historically this may be related to the low levels of water stress in countries where LCA has developed (primarily Europe) and its application to industrial processes that utilise comparatively low volumes of water (Mila i Canals et al. 2009). LCA does however have a strong methodological basis from which to incorporate water usage estimates. LCA is used for assessing resource usage and impacts to humans or the environment, both of which are relevant to water usage. The approaches discussed will present definitions for both assessment of resource usage and, where relevant, impacts from water usage. 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), funds may be characterised as sub-artesian groundwater supplies or dams (exhaustible resources), while flows refer to streams and rivers (non-exhaustible in principle).

Owens (2002) further defined water in terms of in-stream uses (i.e. hydroelectric generation) and off-stream withdrawal, and suggests classifying water by source from surface water or groundwater. Classification of water return or disposition is then suggested, with the options being:

- **Water use –** water is used off-stream and is then 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).

Building on these definitions, Owens presents five water use and water depletion indicators:

- In-stream water use indicator (i.e. the quantity of water used for hydro-electric power generation).
- In-stream water consumption indicator (i.e. evaporative losses from storages and canals in excess of unrestricted river losses).
- Off-stream water use indicator (i.e. surface withdrawals from sustainable sources that are returned to the original basins & groundwater withdrawn from sustainably recharged aquifers and returned to surface waters).

- Off-stream water consumption indicator (i.e. evaporative losses and other conveyance losses, and transfers to another river basin).
- Off-stream water depletion indicator (i.e. withdrawals from overdrawn, unreplenished groundwater sources.

For agriculture, most extracted water represents a consumptive use, as it will be either evaporated, transpired, lost in conveyance or incorporated into a product and removed from the catchment. Water depletion may also be relevant for agricultural systems that withdraw water from the Great Artesian Basin (GAB), which may be classified as an un-replenished source. The methodology presented by Owens is considered foundational in the field of LCA.

Owens (2002) also presents a range of potential indicators for water quality, but does not detail impact categories for human health or ecosystems.

Mila i Canals et al. (2009) have expanded and modified the approach provided by Owens (2002) to provide water characterisation factors for freshwater use. Mila i Canals et al. (2009) integrate the blue and green water terms drawn from the virtual water framework, and propose accounting for these water sources as separate inputs to the life cycle inventory. Water outputs are simplified into two paths, namely *non-evaporative uses* ('water use' under Owens' definition) and *evaporative uses* ('water consumption' under Owens' definition). Mila i Canals et al. (2009) do not consider inter basin transfers as a consumptive use but rather consider this as a change in resource availability between the source and the receiving water basin.

Mila i Canals et al. (2009) identify two main aspects of water that need to be considered, i) water as a resource for humans as competing users, and ii) water as a habitat. Related to these, four impact pathways are identified:

1. Direct water use leading to changes in freshwater availability for humans, leading to changes in human health.
2. Direct water use leading to changes in freshwater availability for ecosystems, leading to effects on ecosystem quality (freshwater ecosystem impact, FEI).
3. Direct groundwater use causing reduced long-term freshwater availability (freshwater depletion, FD).
4. Land use changes leading to changes in the water cycle (infiltration and runoff) leading to changes in freshwater availability for ecosystems, leading to effects on ecosystem quality (FEI).

The association between water use and changes to human health is not straight forward. Other authors have noted that freshwater availability *per se* is not commonly cited as a concern, but access to clean water is (Rijsberman 2006). This author goes on to identify economic status as the primary threat to clean water availability. For these reasons Mila i Canals et al. (2009) suggest omitting this aspect from LCA.

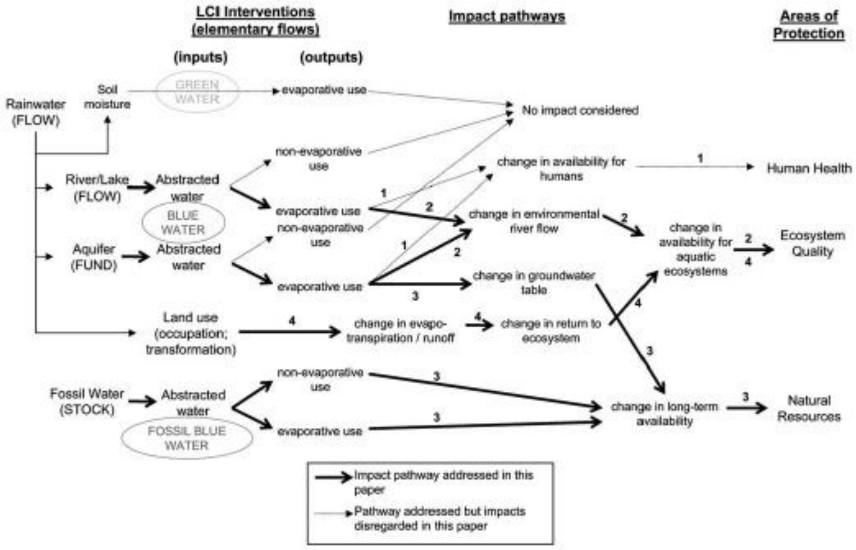

FIGURE 9 – MAIN IMPACT PATHWAYS RELATED TO FRESHWATER USE (MILA I CANALS ET AL. 2009).

Treatment of green water. Green water is included in the framework as an interim to determining blue water requirements for crop irrigation, and to allow comparisons with VW studies. At the impact assessment stage green water and non-evaporative blue water resources are not considered.

To date, their approach has not been demonstrated with published case studies in the literature, though it does have potential for integrating concepts from VW and LCA into a robust method of assessment.

The Australian methodology for agricultural LCA (Harris & Narayanaswamy 2009) provides another alternative to defining and measuring water use. The methodology identifies the following water usage elements in the inventory phase:

- Collected rainwater (treated and untreated).
- Collected surface water (treated and untreated).
- Ground water (treated and untreated).
- Saline and hyper saline water (low quality water for low quality uses).
- Cooling water (treated and untreated) to and from the cooling towers.
- Scheme water (for a centralised water treatment and sewerage works).
- Grey water, potable water (human and animals), irrigation water, etc.
- Treated and untreated storm water run-on and run-off (if captured and used in processes/production activities).

Additionally, water flows associated with feed preparation and incorporation, drinking and service water for animals are to be calculated and included over the entire life span of the animals that contribute to the final product. Water use throughout the life cycle of the product should include, but not be limited to:

- Mining and extraction of raw material (mining operations, dust suppression).
- Manufacturing of materials (e.g. chemicals).
- Irrigation and drinking water.
- Cultivation and processing.

- Heating and cooling (e.g. evaporative losses).
- Transport.
- Evaporation, seepage, drainage etc.

The methodology proposes presenting water use under two definitions, i) the ABS water use definition reported previously in this document (which is roughly equivalent to Blue water), and ii) the following two definitions provided by the National Land and Water Resources Audit (NLWRA):

- Surface water sustainable flow regimes: the volume and pattern of water diversions from a river that include social, economic and environmental needs.
- Groundwater sustainable yield: the volume of water extracted over a specific time frame that should not be exceeded to protect the higher social, environmental and economic uses associated with the aquifer.

The methodology states that the sustainable use of water shall be reported as a percentage of:

- Water removed from rivers as a percentage of sustainable flow regimes.
- Groundwater abstraction as a percentage of sustainable yield.

A weakness of this approach is the lack of comparability with other established methodologies (i.e. Owens 2002) which has been used as a basis for most other water methodology developments in the field of LCA.

Pfister et al. (2009) presents an approach that integrates virtual water measures with LCA, though the attention is focussed solely on blue water use for impact assessment. Impact assessment is carried out by use of a regionalised water stress measure, with a new midpoint category 'water deprivation'. Water deprivation is a measure of the water use (abstracted and evaporative water use, or 'water consumption') related to the degree of water stress within a catchment. The water stress index (WSI) is a measure of the balance of freshwater withdrawals to hydrological availability. Moderate and severe water stress occurs above a threshold of 20 and 40% respectively.

Pfister et al. (2009) use estimates of virtual blue water use for crop production available from global inventories. These are readily available, albeit limited in their accuracy. Using these water use data, water deprivation is measured using the water stress index for the catchment in which production occurs. This provides an indication of the affect that production of a given product is having on actual water stress, rather than simply determining the consumptive water use.

As an example of this methodology, Pfister et al. (2009) present a case study of global cotton production. They show, for example, that although consumptive water use for cotton in Australia (3.92 m^3/kg) is lower than water use in Mali (4.07 m^3/kg), the water deprivation in Australia (1.42 m^3/kg) is higher than Mali (0.99 m^3/kg). This shows the ability of the method to provide information on catchment specific impacts as opposed to simply estimating total volumes of water used. As such this is a major advancement in freshwater impact categories.

Pfister et al. (2009) identify the need for further development of indicators that are able to assess changes in green water flows from production systems.

Collaborative work between Pfister and CSIRO is moving towards generation of a 'stress weighted water use index' (Brad Ridoutt, pers. comm.) which may be a useful method when developed. However, this method has not been presented in the literature as yet.

5.4 Preferred approach to water usage

Estimation of water use in this project will follow an inventory approach that is broadly consistent with Owens (2002) and Harris & Narayanaswamy (2009). Additionally, ‘green’ water has been assessed for the crop growth phase, based on literature values for crop water use and yield. Results will be presented as blue and green water usage, but will not be assigned to impact categories until further agreement is reached for Australian LCA research.

6 Methodology

6.1 Goal and scope

6.1.1 Goal

Goal definition covers the intended application and target audience for the feedlot LCA study. Through consultation with the industry, the following applications were identified:

- Produce a first-order estimate of the resource usage (energy and water) and GHG emissions per kg HSCW gain in the feedlot sector including the feed grain component.
- Allow prioritisation of subsequent research and data collection effort to those areas where the emissions are highest per kg HSCW gain.
- Produce a sensitivity analysis of the parameters in the model.

The target audience for this feedlot LCA is Meat & Livestock Australia (MLA) (industry research funding body) and Australian Lot Feeders' Association (ALFA) (industry body). Due to the preliminary nature of the work, this LCA is not intended for distribution to the general public, Australian beef producers or government agencies.

6.1.2 Scope

The scope of the LCA covers the definition of the functional unit, the proposed system boundary, and data quality requirements. A critical review of the LCA is not included in this study.

6.1.2.1 Functional unit

The functional unit was developed in alignment with the goal and scope of the project, and with reference to previous research and methodology development (i.e. Peters et al. 2009a, Harris and Narayanaswamy 2009). The primary functional unit selected is:

1 kg of liveweight (LWT) gain at the feedlot from point of induction to immediately prior to transport for slaughter.

This represents a 'gate-to-gate' assessment of the feedlot, excluding upstream and downstream processes. A second functional unit was used to provide context for the results (based on literature values for upstream and downstream supply chain sectors). This is:

1 kg of beef (HSCW) at the meat processor docking gate.

6.1.2.2 System boundary

In line with the goal and scope of the project, the system boundary focussed on the feedlot as a sub-set of the beef supply chain. The assessment includes inputs and outputs associated with the feedlot system. Inputs and outputs associated with other sub-sets of the supply chain (upstream livestock production, downstream slaughter) are included in the second functional unit for context but are not the focus of the study. The system, showing the boundary for the current project and the previous MLA study (FLOT.328 – Davis & Watts 2006) is shown in Figure 10.

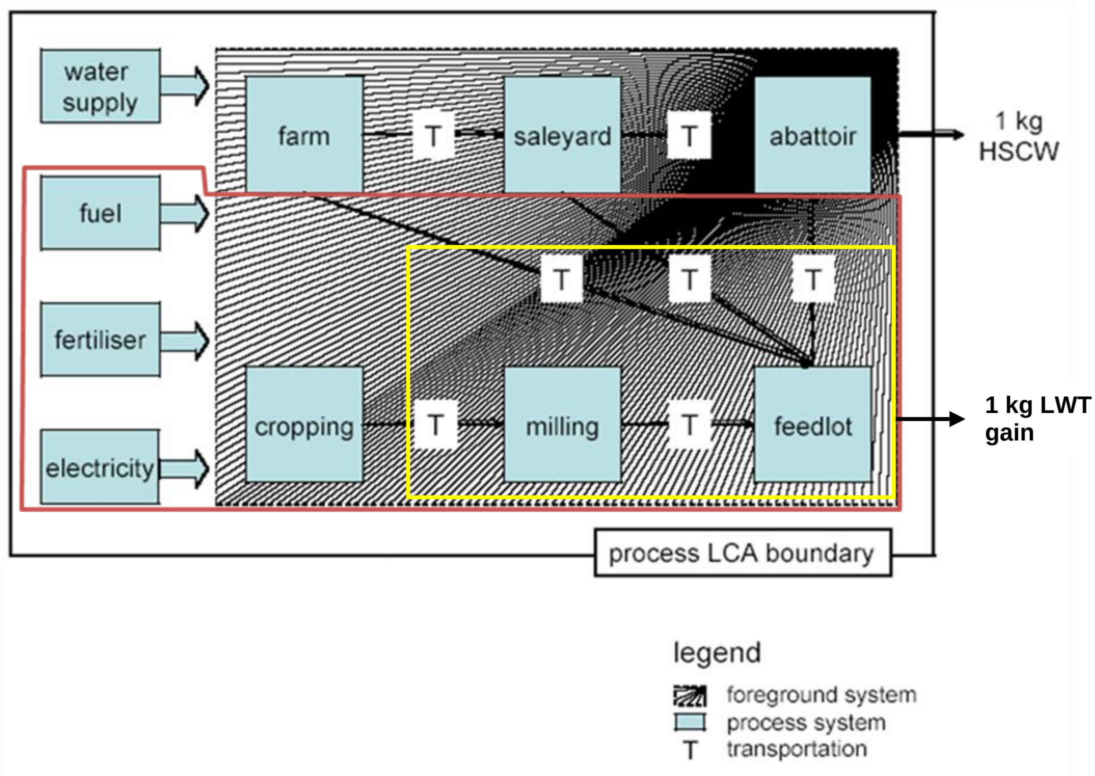

FIGURE 10 – SYSTEM BOUNDARY FOR CURRENT STUDY (RED) AND PREVIOUS LCA STUDY (YELLOW)

6.1.2.3 Data requirements

The data requirements are determined by the goal and scope of the project. In order to provide some comparison between different Australian beef production systems, two feedlots were selected where cattle are fed for either the domestic (short-fed) or export (long-fed) markets. In line with the focus of the project, detailed data were collected at the feedlot site, while literature values were used for other sectors of the supply chain that were included for context.

The foreground data from each supply chain are considered confidential, and data presented in this report are standardised on a per-unit basis so as not to identify the data providers.

6.2 Supply chain description

6.2.1 Feedlots

Data were collected from two feedlots representing different beef supply chains. Feedlot 1 feeds cattle exclusively for the domestic market. The feedlot has an on-site feed mill (tempering system) and produces some grain, hay and silage on-site for use in the ration. Cattle performance details are provided in Table 9.

Feedlot 2 feeds cattle for the long-fed export market. The feedlot has an on-site feed mill (steam flaking) and produces grain, hay and silage on-site for use in the ration. Cattle performance details are provided in Table 9.

TABLE 9 – CATTLE PERFORMANCE DETAILS FOR FEEDLOT 1 AND 2

Parameter	Feedlot 1	Feedlot 2
Avg. Entry Weight (kg)	360	440
Avg. Daily Gain (kg/hd/day)	1.7	0.95
Avg. Total Days on Feed	63	330

The supply chains were extended using generic upstream cattle supply from a Queensland grazing system (adjusted for the difference in entry liveweight between the two feedlots). Meat processing was included using generic processing data which were calculated 'per kilogram of HSCW processed'. These data did not differ for the two supply chains, hence differences in whole supply chain results relate primarily to the feedlot sector.

The supply chains are described as 'supply chain 1' for the short fed system, and 'supply chain 2' for the long fed system.

6.3 Data collection

In line with the goal and scope of the project, data were collected and collated from previously completed projects in most cases.

The Life Cycle Inventory (LCI) was developed using data from previous MLA projects, unpublished FSA Consulting data, literature sources and data from publically available LCA databases such as AUSTLCI and Ecoinvent. In some areas, data gaps were filled by collecting primary data from the feedlots assessed. These are detailed in the following sections.

6.3.1 Feedlot data

Feedlot water and energy data were collected in a previously funded MLA project (B.FLT.0339 – Davis et al. 2008) for eight Australian feedlots. This project did not include GHG emissions data, and did not include some water uses, particularly those associated with the production of feed grain used in the feedlot ration. Additional data modelled for the current project are contained in chapter 7.

6.3.2 Meat processing data

Meat processing data were sourced from MLA (2002) (energy usage, waste stream characteristics) and from data collected at one Queensland meat processing plant (water usage). Waste stream methane was estimated using the DCC methodology (DCC 2007b). This methodology has been recently reviewed for MLA by Wiedemann et al. (2010a) and is not detailed here.

6.3.3 Upstream processes

There are many upstream processes that relate to inputs used by the feedlot. These are associated with energy used (upstream energy supply), commodities used in the ration and the supply of feeder cattle.

Most upstream processes rely on data from LCA databases such as AUSTLCI (where available) and Ecoinvent. However, commodities used in the feedlot ration and production of feeder cattle are inadequately detailed in these inventories. Hence, feed data were modelled by FSA Consulting for feed ration components (FSA Consulting unpublished) based on average yields (taken from the ABS), average crop inputs (taken from NSW DPI gross margins) and local knowledge. The approach used and primary data for most ration components have been reported in Wiedemann et al. (2010b). As with this study, a simplified ration was developed for each feedlot, thereby reducing the number of inputs by substituting variable cereal and protein inputs for the marginal cereal or protein input.

Upstream cattle production (the production of feeder steers) were modelled using the livestock GHG methodology provided by the DCC (2007a). Water and energy usage data for these livestock were modelled based on data from Peters et al. (2009a, b) and from expert knowledge. One difference to the method applied by Peters et al. (2009a, b) was the differentiation of water use into ‘blue’ and ‘green’ water.

6.4 Allocation

Allocation in LCA required where a system produces more than one product. The production of feedlot beef through to carcasses at the meat processing plant leads to the generation of slaughter co-products (i.e. hides, meat meal etc).

Slaughter by-products were allocated using a mass allocation process. Mass allocation divides impacts evenly based on the mass of each product generated. As slaughtering involves some losses and the generation of wastes, the mass of products from a live animal is less than the mass of liveweight. Proportions of each product were taken from MLA (2002).

The allocation process used during meat processing was not felt to be an issue of particular relevance to this project because of the specific goals relating to feedlot emissions. However, allocation at the meat processing plant is relevant when comparing to other literature.

The process applied in this study will cause results to differ from some previous research (i.e. Davis et al. 2008) where other allocation was done entirely to the meat product and to Peters et al. (2010), where the allocation of impacts to wastes was not entirely clear.

TABLE 10 – ALTERNATIVE APPROACHES FOR THE ALLOCATION OF ENVIRONMENTAL IMPACTS TO BEEF CO-PRODUCTS AT THE POINT OF SLAUGHTER

Reference	Functional Unit	Allocation process	Example of allocation process to HSCW			Inclusions / exclusions
			GWP – CO_2-e / kg LWT	Allocation factor	GWP – CO_2-e / kg HSCW	
Davis & Watts 2006, Davis et al. 2008	kg of HSCW gain	All impacts allocated to beef	10	1 / 0.55[1]	18.2	Excludes upstream (breeding) and downstream (meat processing) impacts
Peters et al. 2010 – table 2	kg of HSCW	Comparisons made on the basis of un-allocated impacts	10	1 / 0.53[2]	18.2	Includes upstream and downstream impacts
This study	kg of HSCW	Mass allocation to saleable products / by-products	10	1 / 0.799[3]	12.5	Includes upstream and downstream impacts

[1] This means burdens are equally divided to all products without differentiation, leading to the same emissions per kg HSCW as per kg LWT. This was not clearly elaborated in the paper or in the final report. [2] Average dressing percentage. [3] Yield of saleable by-products including HSCW and minor products such as tallow and meat meal, excluding wastes and carcass losses.

The main problem associated with mass allocation (the primary method used in these studies) relates to the disproportionate allocation of impacts to low quality by-products such as meat meal and tallow. This can be rectified by using an economic allocation process, where a higher proportion of the burden is allocated to meat.

6.5 Impact assessment

Impact assessment was conducted using the indicators GWP and PE. Impact assessment was not conducted for water usage, and data presented are 'water used', differentiated into blue and green water. Impact assessment was done using Simapro 7.1.

7 Life cycle inventory

Additional data collected or modelled as part of the inventory for this project included feedlot GHG emissions and some feedlot water uses.

7.1 Modelling enteric GHG

Enteric methane was modelled using the DCC (2007a) 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 is as follows:

$$Y \text{ (MJ } CH_4\text{/head/day)} = 3.406 + 0.510 \times SR + 1.736 \times H + 2.648 \times C \qquad \textit{(Eqn 1)}$$

Where:

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

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

Hence:

SR = (I x Pgrain x SRgrain) + (I x Pconc x SRconc) + (I x Pgrass x SRgrass) + (I x Plegume x SRlegume)

H = (I x Pgrain x Hgrain) + (I x Pconc x Hconc) + (I x Pgrass x Hgrass) +(I x Plegume x Hlegume)

C = (I x Pgrain x Cgrain) + (I x Pconc x Cconc) + (I x Pgrass x Cgrass) + (I x Plegume x Clegume)

Where:

I = intake (kg/day)
Pgrain = proportion of grains in feed
Pconc = proportion of concentrates in feed
Pgrass = proportion of grasses in feed
Plegume = proportion of legumes in feed
SRgrain = soluble residue content of grain
SRconc = soluble residue content of other concentrates
SRgrass = soluble residue content of grasses
SRlegume = soluble residue content of legumes
Hgrain = hemicellulose content of grain
Hconc = hemicellulose content of concentrates
Hgrass = hemicellulose content of grasses
Hlegume = hemicellulose content of legumes
Cgrain = cellulose content of grain
Cconc = cellulose content of concentrates
Cgrass = cellulose content of grasses
Clegume = cellulose content of legumes

The total daily production of methane, M (kg methane/head/day) is thus:

$M = Y / F$ *(Eqn 2)*

Where:

F = 55.22 MJ/kg CH_4

The DCC provide default values for daily feed intake and feed properties for Australian feedlot cattle, and these have been used to generate a 'DCC methodology scenario'. For the two feedlots under investigation, actual data were available and were substituted into the equation for the 'standard' run, as these are more accurate of the systems under investigation than the default assumptions by the DCC. Key differences between the DCC 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 11 – ENTERIC METHANE ASSUMPTIONS FOR FEEDLOT 1 (DOMESTIC CATTLE)

		DCC (2007a)	Actual data – feedlot 1
Daily Intake (assume DMI)	(kg/day)	9.8	10.12
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

[1] forage hay / silage classified under grasses

TABLE 12 – ENTERIC METHANE ASSUMPTIONS FOR FEEDLOT 2 (LONG-FED CATTLE)

		DCC (2007a)	Actual data – feedlot 2
Daily Intake (assume DMI)	(kg/day)	11.0	8.6
Proportion of grains in feed	(%)	0.779	0.684
Proportion of concentrates in feed	(%)	0.048	0.022
Proportion of grasses in feed[1]	(%)	0.138	0.294
Proportion of legumes in feed	(%)	0.035	0.001

[1] forage hay / silage classified under grasses

Default values for soluble residue, hemicelluloses and cellulose content of each feed category were taken from the DCC (2007a).

7.2 Modelling manure GHG

7.2.1 Department of Climate Change method

The default method for estimating manure emissions from feedlot cattle is provided by the DCC (2007a). This method is summarised in the following sections for the two emission sources, manure methane and nitrous oxide.

7.2.1.1 Manure methane emissions

The rate of methane emissions depends upon the volatile solids content of the manure and the manure management system. The estimation of methane emissions from manure is based on an estimate of the volatile solids content of manure, taking into consideration the emissions potential (B_o) and the yield for a given Manure Management System (expressed as the Manure Conversion Factor - MCF). Casada and Safley (1990) developed a method for estimating methane releases from manure by making certain assumptions about the percentage of ultimate methane yield (B_o in m^3 CH_4/kg VS) that could be expected by different manure management systems. B_o is the ultimate methane yield of an anaerobically digested material. The percentage of B_o that is achieved by different manure management systems is the MCF. B_o varies with animal species and diet. Table 13 gives 'standard' values of B_o for various livestock types but it must be emphasised that these are averages and can vary substantially depending on diet. In general, ruminants have a low B_o for their manure presumably because most of the methane potential is extracted as enteric methane during fermentation in the rumen. Ruminant manures have a higher proportion of remaining carbohydrates that are difficult to break down, such as cellulose and lignin.

TABLE 13 – METHANE-PRODUCTION POTENTIAL, B_o, FOR DIFFERENT INTENSIVE LIVESTOCK

Livestock Category	Sub-group	Liveweight (kg)	Manure (kg DM/hd/day)	VS (kg/hd/day)	B_o (m^3 CH_4/kg VS)
Dairy Cattle	Mature females	500	3.77	3.47	0.24
Beef Cattle	Mature females	400	3.91	3.60	0.17
	Mature Males	450	3.38	3.11	0.17
	Young	200	2.41	2.21	0.17
Pigs	Average	82	0.51	0.43	0.45

Source: IPCC (1997a) for Oceania and developed countries.

The MCF provides an estimate of the portion of the methane-producing potential of waste that is achieved (IPCC 1997b). Different waste management systems and climatic conditions affect the methane-producing potential of waste. Manure managed as a liquid under hot conditions has higher methane formation and emissions and hence a high MCF value. Manure managed as a dry material in cold climates does not readily produce methane and consequently has a lower MCF.

The DCC (2007a) method of estimating emissions from manure is as follows, beginning with estimation of volatile solids (VS kg/head/day):

$$VS = I \times (1 - DMD) \times (1 - A) \qquad \text{(Eqn 4)}$$

Where:

I = Dry matter intake.

DMD = digestibility expressed as a fraction (assumed to be 80%).

A = ash content expressed as a fraction (assumed to be 8% of faecal DM).

Volatile solids are calculated using standard figures for dry matter intake and ration digestibility and have been developed using BEEFBAL (Queensland Department of Primary Industries and Fisheries 2005).

Following VS estimation, methane production from faeces, M (kg/head/day) is calculated as:

$$M = VS \times B_o \times MCF \times \rho \qquad \text{(Eqn 5)}$$

Where:

B_o = emissions potential (0.17m^3 CH_4/kg VS)

MCF = methane conversion factor (Drylot MCF values for 'warm' regions such as Queensland and the Northern Territory =5%, MCF values for 'temperate' regions (for all other States) = 1.5%.

ρ = density of methane (0.662 kg/m^3)

The DCC simplify manure management at feedlots into a single manure management system (drylot) and therefore consider only point of emission (presumably the feedpad). Hence, any *losses occurring from the effluent pond, sedimentation basin or effluent irrigation were not considered in the DCC scenario.*

7.2.1.2 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, resulting in harmful emissions to the environment. 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 storage within the animal. The following algorithms are used to calculate crude protein input (CPI) and storage (NR).

CPI (kg/head/day) = NI x 6.25 *(Eqn 7)*

Where

NI = nitrogen intake (kg/day)
6.25 = factor for converting nitrogen into crude protein

NI is calculated from the nitrogen concentration of different dietary components and the proportion of these components in the ration, using the same ration breakdown as was used for enteric methane estimation. This is detailed in the following formula:

$NI = (I \times P_{grain} \times N_{grain}) + (I \times P_{conc} \times N_{conc}) + (I \times P_{grass} \times N_{grass}) + (I \times P_{legume} \times N_{legume})$

Where:
N_{grain} = nitrogen content of grain
N_{conc} = nitrogen content of other concentrates portion of the diet
N_{grass} = nitrogen content of grasses portion of the diet
N_{legume} = nitrogen content of legumes portion of the diet

The methodology for estimating nitrogen excretion in manure, F (kg/head/day) is based on the indigestible fraction of the undegraded protein from solid feed and the microbial crude protein, plus the endogenous faecal protein. This methodology takes a mass balance approach where N output = N input - N storage. The total N output is then split into urinary and faecal components.

The nitrogen excreted in faeces (F kg/head/day) is calculated as:

$F = \{0.3(CPI \times (1-[(DMD+10)/100])) + 0.105(ME \times I \times 0.008) + 0.0152 \times I\} / 6.25$ *(Eqn 8)*

Where:

DMD = digestibility expressed as a percentage (assumed to be 80%)
ME = metabolisable energy (MJ/kg DM) is calculated:

$ME = 0.1604 \times DMD - 1.037$ (*Eqn 9*)

I = feed intake (kg/day)

The amount of nitrogen that is retained by the body, NR (kg/head/day) is calculated as the amount of nitrogen retained as body tissue such that:

$$NR = \{[0.212-0.008(L - 2) - \{(0.140-0.008(L - 2)) / (1+exp(-6(Z - 0.4)))\}] \times (LWG \times 0.92)\}/6.25 \quad \text{--- (Eqn 10)}$$

Where:

L = Relative intake, which is feed intake divided by the intake require for maintenance.

Z = Relative size (liveweight/standard reference weight)

LWG = Liveweight gain

Nitrogen excreted in urine (U kg/head/day) is calculated by subtracting NR, F and dermal protein loss from the nitrogen intake such that:

$$U = (CPI / 6.25) - NR - F - [(1.1 \times 10^{-4} \times W^{0.75})/6.25] \quad \text{(Eqn 11)}$$

Where:

W = Liveweight

The total annual faecal (AF) and urinary (AU) nitrogen excreted is then calculated by:

$$AF = (N \times F \times 365) \times 10^{-6}$$

$$AU = (N \times U \times 365) \times 10^{-6}$$

Where:

F = Eqn 8
N = the annual equivalent number of feedlot cattle.
U= Eqn 11

Once excreted nitrogen has been estimated, losses of nitrous oxide and ammonia can be calculated.

Feedpad emissions

The total emissions of nitrous oxide from the feedpad (designated ‘Drylot’ by the DCC) are calculated as follows:

$$Faecal_{MMS} = (AF \times MMS \times EF_{(MMS)} \times 44/28) \quad \text{(Eqn 12)}$$

$$Urine_{MMS} = (AU \times MMS \times EF_{(MMS)} \times 44/28) \quad \text{(Eqn 13)}$$

$$Total_{MMS} = (Faecal_{MMS} + Urine_{MMS}) \quad \text{(Eqn 14)}$$

Where:

MMS = the fraction of the annual nitrogen excreted (AU + AF) that is managed in the different manure management systems. It is assumed that with feedlot cattle all manure is dry packed (MMS = 4).

$EF_{(MMS)}$ = emission factor (N_2O-N kg/ N excreted) for the different manure management systems.

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

Emissions of ammonia from the feedpad are calculated as 30% of excreted N.

The DCC (2007a) do not explicitly identify partitioning of nitrogen between solid and liquid (effluent) storage, nor are equations or emission factors supplied for losses from either solid or liquid storage. Hence, losses occurring from the manure stockpile, sedimentation basin and effluent pond are not considered in the DCC scenario.

Manure application losses

The DCC (2007a) estimate that further losses of nitrous oxide occur following application of solid manure. Emission estimation relies on an estimate of applied nitrogen, which is estimated as excreted N less losses of nitrous oxide N (N_2O-N) and ammonia N (NH_3-N) as calculated above.

Once the mass of N available for land application is determined, emissions are calculated as 1% of applied N.

Losses from effluent application are not identified by the DCC methodology and were not considered in the DCC scenario.

Indirect nitrous oxide emissions

The DCC identify further nitrous oxide emissions associated with feedlots via the volatilisation and deposition of ammonia nitrogen from the feedlot. This nitrogen is subsequently available for re-volatilisation as nitrous oxide. Ammonia nitrogen losses are estimated at 30% of excreted N. Of this, 1% is re-volatilised as nitrous oxide. No further losses are identified.

7.2.2 Mass balance method

The DCC methods for estimating manure methane and nitrous oxide were designed to enable a national inventory to be developed. The methods for estimating VS and excreted are based on Australian mass balance research (van Sliedregt et al. 2000) and using the waste estimation mass balance program BEEFBAL (QPIF 2004b). Considering this, a reasonable alternative to the DCC methods *for estimation of VS and excreted N* is to use mass balance principles based on actual feed intake and ration components.

The BEEFBAL program enables the estimation of excreted VS and nitrogen 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 DCC (2007a) in section 7.2.1. The program accounts for partitioning between the effluent pond and solid storage, and traces nitrogen through to land application as effluent or manure.

Consequently, BEEFBAL is a more comprehensive basis for estimating GHG from the whole manure management system at the feedlot. However, as the program was not developed with GHG in mind, several ‘gaps’ exist in the program, and many of the formulas for partitioning and losses of VS and N are imprecise.

None-the-less, 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, and this should be seen as the methodological framework for all future manure GHG research.

For indicative purposes, a theoretical mass balance was developed for the feedlot system beginning with feed intake and tracing VS and nitrogen through each major stage of the feedlot to the point of application. This was based on the mass balance established for the review of ammonia emissions for the Australian feedlot industry (FSA Consulting 2006) and other literature sources. Where no information was available, ‘best-estimate’ values were used.

Emissions of methane and nitrous oxide used emission formulas and factors supplied by the DCC (2007a) generally. For additional emission pathways, emission factors from the IPCC (Dong et al. 2006, De Klein et al. 2006) or from literature reviewed in Redding (2010) were used.

Figure 11 shows the theoretical mass balance for VS for Australian feedlots. The theoretical mass balance of nitrogen, including partitioning and loss pathways, is shown in Figure 12.

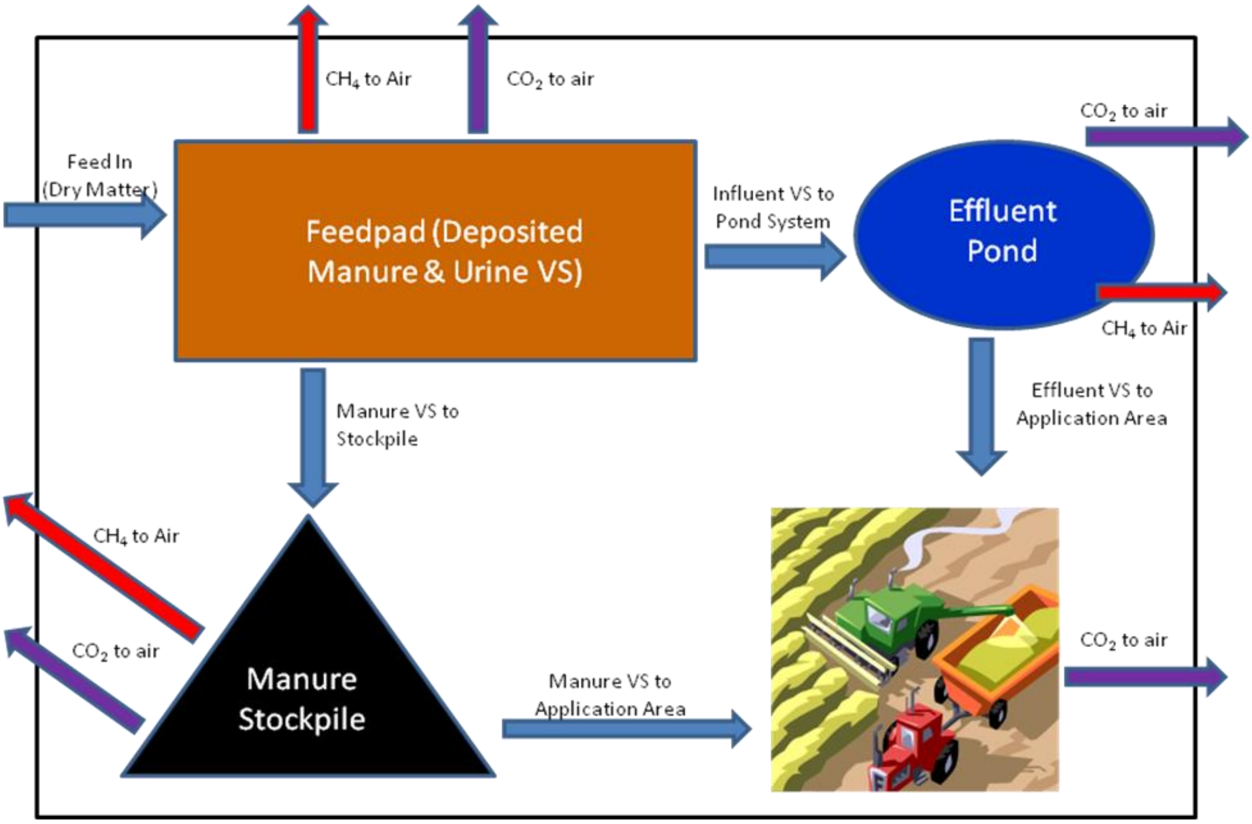

FIGURE 11 – THEORETICAL MASS BALANCE FOR EXCRETED VOLATILE SOLIDS IN AUSTRALIAN FEEDLOTS

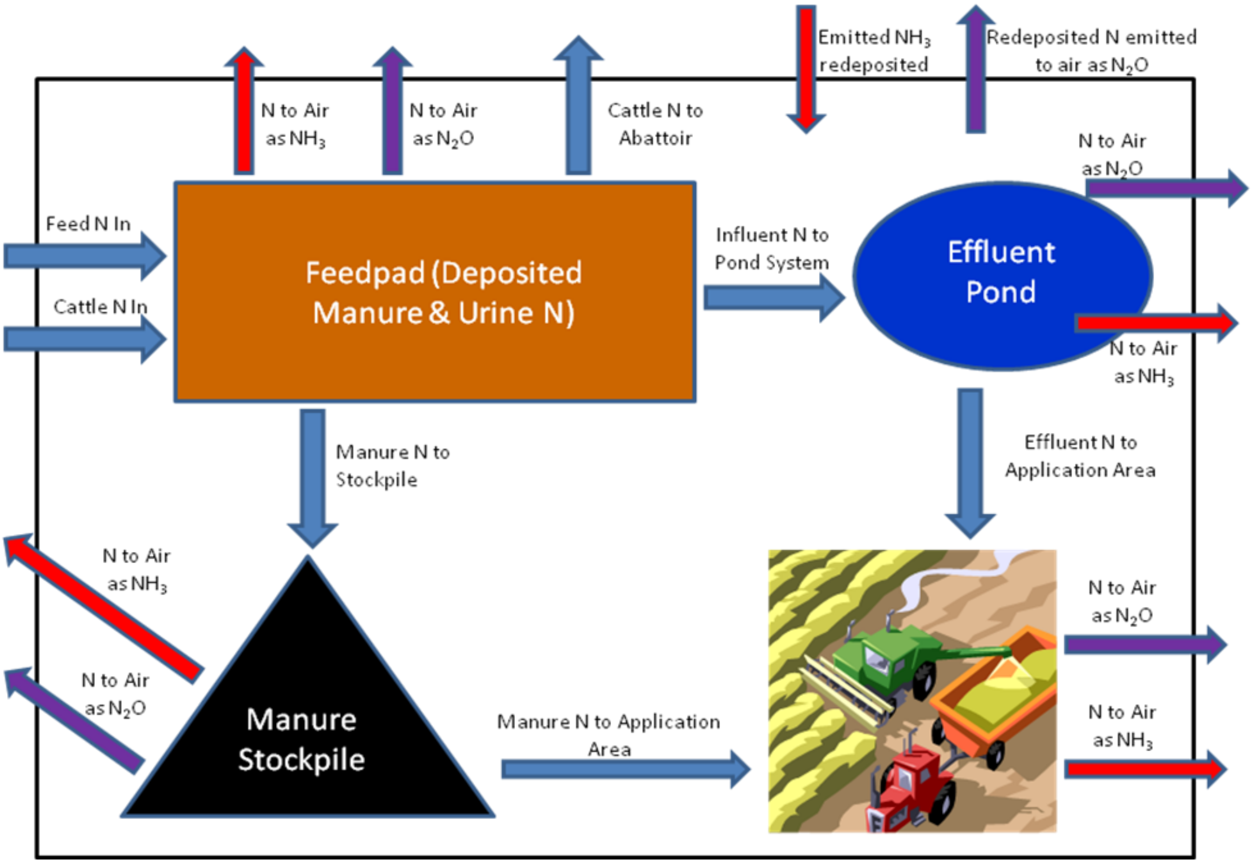

FIGURE 12 – THEORETICAL MASS BALANCE FOR EXCRETED NITROGEN IN AUSTRALIAN FEEDLOTS

Literature and default values used for volatile solids, excretion and partitioning between the solid and liquid storage systems and reuse are provided in Table 14.

TABLE 14 – LITERATURE AND DEFAULT VALUES USED FOR EXCRETION AND HARVESTING OF VS (KG/HEAD/YR)

Emission source	Number	VS Excretion Rates (kg/head/yr)		Reference
		Value	Range	
Excreted	Literature review	-	602	Gilbertson et al 1974 in Kissinger et al. 2007
	Literature review	- -	635 – 923	NRCS 1992a in Kissinger et al. 2007
	Literature review		475 – 1241	Lorimor et al. 2000 in Kissinger et al. 2007
	ASABE Standards – based on dry matter intake and feed dry matter digestibility	-	597 (SF) – 598 (LF)	ASABE 2005
	Based on feed intake of 9.8 kg DM/hd/d (short-fed) & 11 kg DM/hd/d, DMD = 80% and ash content of diet = 8%.	-	659 (SF) – 739 (LF)	DCC 2007a
	Based on gross digestible and urinary energy and ash content of diet	-	640 (LF) – 750 (SF)	IPCC 2006
	Default used based on intake, dry matter digestibility and ash content of feed ingredients – DMDAMP	***-***	***822 (LF) – 891 (SF)***	***Beefbal***
Harvested VS	Values from 6 Nebraska feedlots	548	37 – 1022	Kissinger et al. 2007
	Summer vs winter data for 18 manure harvesting experiments in Nebraska	2570	562 – 3372	Kissinger et al. 2006
			484 (SF) – 554 (LF)	DCC 2007a
			470 (LF)-551 (SF)	IPCC 2006
	Default based on 2% VS to pond, 25% VS lost from pad as CH_4 and methane loss based on DCC 2007a methodology	***-***	***604 (LF) – 655 (SF)***	***Beefbal***
VS to pond in runoff			9.9 (SF) – 11.1 (LF)	DCC 2007a
			9.6 (LF)-11.2 (SF)	IPCC 2006
	Default based on 2% VS to pond from Literature review		***12.2 (LF) – 13.4 (SF)***	***Beefbal***

Note: LF and SF – denotes calculated values for long-fed and short-fed scenarios respectively.

Literature and default values used for volatile solids losses as methane and carbon dioxide from the feedpad and stockpiled manure and sequestered carbon from application are provided in Table 15.

TABLE 15 – LITERATURE AND DEFAULT VALUES USED FOR VS LOSSES FROM FEEDPAD AND STOCKPILE AS METHANE AND CARBON DIOXIDE AND SEQUESTERED CARBON FROM APPLICATION (KG/HEAD/YR)

Emission source	Number	**VS Losses (kg/head/yr)**		**Reference**
		Value	**Range**	
Feedpad VS loss as CH_4 Emissions	Based on emission potential of 0.17 m3 CH_4/kg VS, density of methane of 0.662 kg/m^3 and MCF values of 5% for warm regions and 1.5% for temperate regions.		3.8 (SF) – 4.2 (LF)	DCC 2007
	Based on emission potential of 0.17 m3 CH4/kg VS, density of methane of 0.662 kg/m^3 and MCF values of 2%.		1.5 (LF) – 1.7 (SF)	IPCC 2006
	Default used – DCC 2007a		***4.6 (LF) – 5.0 (SF)***	***Beefbal***
Feedpad VS loss as CO_2 Emissions	Based on mass balance – Vs excreted minus CH_4 loss = CO_2 loss	-	161 (SF) - 181 (LF)	DCC 2007a
	Based on mass balance – Vs excreted minus CH_4 loss = CO_2 loss	-	158 (LF) – 186 (SF)	IPCC 2006
	Default based on mass balance – Vs excreted minus CH_4 loss = CO_2 loss	***-***	***206 (LF) – 218 (SF)***	***Beefbal***
Manure stockpile CH_4 Emissions	No values provided for stockpile emissions	-	0.0 (SF) – 0.0 (LF)	DCC 2007a
	Methane conversion factor 80% for uncovered anaerobic ponds	-	0.8 (LF)- 0.9 (SF)	IPCC 2006
	Default based on IPCC 2006	***-***	***1.0 (LF) – 13.4 (SF)***	***Beefbal***
Manure stockpile CO_2 Emissions	From measured data of FSA Consulting – VS content stockpiled manure = 55%	-	151(SF)-170(LF)	DCC 2007a
	As above	-	146 (LF) - 171 (SF)	IPCC 2006
	As above	***-***	***186 (LF) – 203 (SF)***	***Beefbal***
Application area – Sequestered carbon	High value	-	6 (SF) – 19 (LF)	Redding 2010
	Low value	-	93 (LF) - 100 (SF)	Redding 2010
	Default used based on best estimate from Redding 2010 with manure applications from 7 – 48 t/ha/yr	***-***	***6 (LF) - 20(SF)***	***Beefbal***

Note: LF and SF – denotes calculated values for long-fed and short-fed scenarios respectively.

Table 16 to Table 21 provide literature and default values used for this study of the partitioning and gaseous emissions (ammonia and nitrous oxide) of nitrogen in a feedlot production system from feed intake through to manure and effluent application.

Literature values and default values used for this study for nitrogen excretion rates as a percentage of feed intake are provided in Table 16.

TABLE 16 – NITROGEN EXCRETION RATES AS A PERCENTAGE OF N INTAKE

Details of literature	Percentage of N intake		Reference
	Value	Range	
Varying P diet composition of diet	-	79.6 - 86.2	Sinclair 1997
Varying bran composition of diet	-	90.2 - 91.2	Erikson et al 2002
Varying bran composition of diet	-	83.8 - 85.2	Farran et al. 2004
Varying WDG composition of diet	-	82.2 - 88.5	Luebbe et al. 2008
	-	83.8 - 86.6	Luebbe et al. 2008
Based on Intake minus retention, where intake is0.5kg/d/1000kg lwt & retention is 7% of intake	93.0	-	IPCC 2006
Default used – Mass balance based on Feed Intake – retention (0.027 for starter cattle & 0.024 for finisher cattle)	**-**	***91.1-92.2 for this study***	***Beefbal***

Literature and default values used for this study for nitrogen emissions of ammonia and nitrous oxide and partitioning to the pond system and manure stockpile from the feedpad as a percentage of nitrogen excreted are provided in Table 17.

TABLE 17 – NITROGEN IN RUNOFF AND HARVESTED MANURE AND EMISSIONS OF AMMONIA AND NITROUS OXIDE FROM THE FEEDPAD AS A PERCENTAGE OF N EXCRETED

Emission source	Details of literature	Percentage of N Excreted		Reference
		Value	Range	
Feedpad NH_3–N emissions			50.0 - 55.0	Flesch et al. 2007
	6 to 12 months cleaning intervals		57.0 - 67.0	Biermann et al. 1999
	18 harvesting Experiments - Nebraska		47.0 - 69.0	Kissinger et al. 2006a
	Included bran treatments in diets		25.2 - 47.9	Farran et al. 2004
	Varying pen cleaning frequency		55.5 - 78.4	Wilson et al. 2004
	10 week study – Texas, USA		62.0 - 64.0	Todd et al. 2006
	2 month study – Texas, USA		63.0 - 65.0	Flesch et al. 2007
	Review of literature for NPI Review	80.0	-	FSA Consulting 2006
	Suggested values	30.0	-	IPCC 2006
	Suggested values	30.0	20.0 - 50.0	DCC 2007a
	Default used – based on literature and measured harvested manure N values	***70.0***	-	***BeefBal***
Feedpad N_2O-N emissions	Suggested values	2.0	1.0 - 4.0	IPCC 2006
	Suggested value	2.0		DCC 2007a
	Default used-from DCC and IPCC	***2.0***		***Beefbal***
N to Effluent pond	18 harvesting experiments in Nebraska		0.05 - 5.0	Kissinger et al. 2006a
	High value was large runoff event		4.6 - 19.4	Biermann et al. 1999
	Default used – based on N balance estimates	***2.0***		***BeefBal***
N to manure solid storage	***Defaults calculated for this study – based on mass balance principles***	**26.0**	**-**	***Beefbal***

Literature and default values used for this study for nitrogen partitioning and emissions of ammonia and nitrous oxide for the effluent treatment and storage system as a percentage of nitrogen entering the pond system are provided in Table 18.

TABLE 18 – NITROGEN PARTITIONING AND EMISSIONS OF AMMONIA AND NITROUS OXIDE FROM THE EFFLUENT TREATMENT AND STORAGE SYSTEM AS PERCENTAGE OF N ENTERING POND SYSTEM

Emission source	Details of literature	Percentage of N to Pond		Reference
		Value	Range	
N to Pond Sludge from runoff				*FSA Consulting 2006*
	Default used – based on FSA Consulting 2006	***30.0***		***Beefbal***
Pond NH_3-N emissions	Values from dairy ponds as no data for beef feedlots	35.0	20 – 80	IPCC 2006
	Value from dairy ponds as no data for beef feedlots	35.0		DCC 2007
	Review of literature for NPI Review	35.0		*FSA Consulting 2006*
	Default used – based on IPCC, DCC and FSA Consulting 2006 NPI review	***35.0***		***Beefbal***
Pond N_2O-N emissions	Assumes no N_2O emissions from anaerobic ponds	0.00		IPCC 2006
	Value for uncovered anaerobic ponds	0.10		DCC 2007a
	Default used -Value from DCC 2007	***0.10***		***Beefbal***
N irrigated – fraction of excretion	***Defaults calculated for this study – based on mass balance principles***	***34.9***	***-***	***Beefbal***

Literature and default values used for this study for nitrogen emissions as ammonia and nitrous oxide from solid manure storage as a percentage of nitrogen stockpiled are provided in Table 19.

TABLE 19 – NITROGEN EMISSIONS OF AMMONIA AND NITROUS OXIDE FROM STOCKPILED MANURE AS A PERCENTAGE OF STOCKPILED NITROGEN

Emission source	Details of literature	Percentage of N Stockpiled		Reference
		Value	Range	
Storage NH_3–N emissions	Table 10.22 of IPCC 2006	45	10 - 65	IPCC 2006
	Value for dairy – no value for beef cattle feedlot provided	30	-	DCC 2007a
	Review of literature for NPI Review	25	15 - 40	FSA Consulting 2006
	Default used – based on FSA Consulting 2006 NPI Review	***25***	***-***	***Beefbal***
Storage N_2O-N emissions	U.K. straw bedding system stockpile for pig manure	2.6 -	- -	Thorman et al 2007
	Passive aeration versus turning	-	0.62 - 1.7	Hao et al. 2001
	Intensive composting (regular turning)	10.0	5.0 – 20.0	IPCC 2006
	Static piles with forced aeration	0.6	0.3 - 1.2	IPCC 2006
	Passive windrow – infrequent turning	1.0	0.5 – 2.0	IPCC 2006
	Solid storage	0.50	0.27 – 1.0	IPCC 2006
	Default used - from IPCC 2006 for solid storage	***0.5***		***Beefbal***
Manure N to Application area	***Defaults calculated for this study – based on mass balance principles***	***74.5***		***Beefbal***

Literature and default values used for this study for nitrogen emissions of ammonia and nitrous oxide from manure application as a percentage of nitrogen ap[plied are provided in Table 20.

TABLE 20 –NITROGEN EMISSIONS OF AMMONIA AND NITROUS OXIDE FROM MANURE APPLIED AS A PERCENTAGE OF N STOCKPILED

Emission source	Details of literature	Percentage of N Applied		Reference
		Value	Range	
Application NH_3 emissions	Solid cattle manure application	20	8 - 60	Rotz 2004
	Review of literature for NPI Review	-	10 – 30	FSA Consulting 2006
	Default used -from FSA Consulting 2006 for effluent reuse	***15***		
	Default used -from FSA Consulting 2006 for manure application	***20***		
Manure Application N_2O emissions	Solid cattle manure application	-	1 - 4	Rotz 2004
	4 studies from Canada applying 173-510 kg N/ha/yr	0.3	0.2 - 0.4	Lessard et al. 1996
	From Canada – 1 yr study		2.0 – 3.4	Chang et al. 1998
	Reported range from literature review		0.003 - 0.9	Redding 2010
	Table 11.1 – Value for mineral fertilisers, organic amendments and crop residues	1.0	0.3 – 3.0	IPCC 2006
	Generic value of 1.0 chosen despite literature values of manure being lower	1.0	0.04 – 3.3	DCC 2007a
	Default used -from IPCC 2006 & DCC 2007a	***1.0***	***-***	***Beefbal***

Literature and default values used for this study for nitrous oxide emissions from the atmospheric deposition of ammonia as a percentage of nitrogen volatilised from the system are provided in Table 21.

TABLE 21 –NITROUS OXIDE FROM ATMOSPHERIC DEPOSITION OF AMMONIA AS A PERCENTAGE OF NITROGEN VOLATILISED AS AMMONIA FROM THE SYSTEM

Emission source	Details of literature	Percentage of N Applied		Reference
		Value	Range	
N_2O emissions from atmospheric deposition	Table 11.3.	1.0	0.2 – 5.0	IPCC 2006
		1.0	-	DCC 2007a
	Default used –from IPCC 2006 & DCC 2007a	***1.0***	***-***	***Beefbal***

To summarise these literature, the factors used in the present study are shown in Table 22.

TABLE 22 –EMISSION FACTORS USED IN THE PRESENT STUDY FOR MANURE EMISSIONS

Emission source	DCC (2007a) methodology	Theoretical mass balance	Reference for emission factor used in the theoretical mass balance
Storage and Feedpad (N_2O)	2 %	2 %	DCC – feedlot beef
Feedpad (CH_4)	5 % or 1.5%[1]	5 % or 1.5%[1]	DCC – feedlot beef
Feedpad (NH_3)	30 %	70 % +	NPI – FSA Consulting (2006)
Manure Storage (N_2O)	-	0.5%	IPCC default
Manure Storage (CH_4)	-	5 %	IPCC default
Effluent Pond (N_2O)	-	0.1%	DCC – dairy industry
Effluent Pond (CH_4)	-	80 %	DCC – dairy industry
Manure Application (N_2O)	1 %	1 %	DCC – manure application
Effluent Application (N_2O)	-	1 %	DCC – dairy industry
Atmospheric deposition (N_2O)	1 %	0.2 %	IPCC – low value

[1] feedpad methane emission factor varies for northern states (QLD – 5%) and southern states (NSW, VIC = 1.5%).

For the short-fed supply chain scenario, the Beefbal estimations of the amount of nitrogen partitioned throughout the system (excreted, liquid effluent, solid storage, land application losses to air as ammonia and nitrous oxide) is shown in Table 23.

TABLE 23 – THEORETICAL MASS BALANCE ESTIMATES FOR THE PARTITIONING OF NITROGEN (KG/HEAD/YR AND % OF INTAKE) FOR SUPPLY CHAIN 1 (SHORT-FED)

Source or emission of nitrogen	kg/head/yr	% of N Intake
Animal mass balance		
N Intake	95.8	100.0
N in LW gain	7.7	8.1
N in mortalities	0.8	0.79
N Excreted	87.3	91.1
Losses and partitioning on pad		
Volatilised from pad as NH_3	61.1	63.8
Volatilised from pad as N_2O	1.7	1.8
N to Pond	0.5	0.51
Harvested N from Pad	24.0	25.0
Pond partitioning and losses		
N to sludge	0.15	0.15
Volatilised from pond as NH_3	0.12	0.13
Volatilised from pond as N_2O	0.00	0.00
Irrigated from pond	0.22	0.23
Irrigation losses		
Irrigated from pond	0.22	0.23
Volatilised from irrigation area as NH_3	0.03	0.03
Volatilised from irrigation area as N_2O	0.002	0.002
Stockpile losses		
Harvested from Pad	24.0	25.0
Volatilised from stockpile as NH_3	6.0	6.3
Volatilised from stockpile as N_2O	0.12	0.13
Manure application losses		
N applied as manure	17.9	18.6
Volatilised from manure application area as NH_3	4.5	4.7
Volatilised from manure application area as N_2O	0.18	0.19
Indirect emissions from NH_3 Deposition		
Indirect Emissions from N_2O – Pad	0.61	0.64
Indirect Emissions N_2O - Pond	0.001	0.001
Indirect Emissions of N_2O - Stockpile	0.06	0.06
Indirect Emissions of N_2O - Irrigation	0.0003	0.0003
Indirect Emissions of N_2O - Manure Application	0.04	0.05
Total Indirect Emissions of N_2O	*0.72*	*0.75*

7.3 Water usage in ration commodities

Grain and forages used in feedlot rations contain ‘embedded’ water that was used during production. This water is predominantly soil water (derived from rainfall) that is used in evapotranspiration (green water). However, some common feedlot ration commodities are grown with irrigation or partial irrigation. These include cotton seed (a co-product of cotton production) and silage or hay (often grown on site at the feedlot). While these commodities typically make up a small proportion of the ration, the water used for production may be very high, and may contribute greatly to overall water use.

Water usage for grain and forage commodities have not been adequately researched in Australia to date. As part of this study, literature estimates of evapotranspiration requirements of crops in eastern Australia were used to generate green water usage data. Irrigated water usage was estimated from cotton water usage (for cotton seed) and through local knowledge of irrigation requirements for silage production at feedlots in eastern Australia.

Of the two feedlots investigated, one used a higher proportion of commodities from irrigated sources.

7.4 Additional water usage modelling at the feedlot

Davis et al. (2008) collected comprehensive water usage data for feedlots, including both beneficial uses (such as drinking water) and non-beneficial uses (such as evaporation from storages). However, in line with progressive theories of water use in LCA, additional water uses also exist at the feedlot not captured by Davis et al. (2008). The major use not accounted for previously was water captured in the effluent containment ponds.

7.4.1 Effluent water capture

Water can be considered as ‘used’ if it is restricted from entering a natural waterway. This follows the theory that, if water is restricted from entering a waterway it is not available for competing uses, such as downstream irrigation or ‘environmental flows’. Feedlots are required to capture all runoff from the feedlot complex to comply with environmental regulations related to nutrient loss from the site. This activity restricts a proportion of water from contributing to natural runoff and stream flows, hence it is rightfully considered a ‘water use’ attributable to the feedlot.

Additional water use associated with runoff capture was modelled for each feedlot, based on annual rainfall, feedlot area and standard feedlot runoff coefficients.

8 Results

To gain a greater understanding of emissions specific to the feedlot component of the supply chain, a 'gate-to-gate' study was completed that excluded 'upstream' emissions associated with cattle breeding, and downstream emissions associated with meat processing. Results are presented on a 'per kg of liveweight gain' basis. The gate-to-gate assessment included impacts related to direct inputs such as grain used in the feedlot ration which were not previously included in Davis & Watts (2006). It should also be noted that Davis & Watts (2006) reported results 'per kg of HSCW gain' which is an 'unallocated' method that applies all burdens to the carcass weight.

Whole supply chain results (including upstream breeding and backgrounding and downstream meat processing) are presented for context in the discussion section.

8.1 GHG emissions

The project aimed to generate 'first order' estimates of GHG emissions from two feedlots as a gate to gate assessment. Many gaps exist in the data and methodology for estimating specific GHG sources from livestock, hence these estimates are preliminary in nature, representing the 'state of the science' to date. Greenhouse gases were aggregated using GWP as an indicator.

Total emissions for supply chain 1 (short fed) were 7.5 kg CO_2-e / kg LWT gain. For supply chain 2 (long fed) emissions were 11.3 kg CO_2-e / kg LWT gain. Major contributions to emissions at the feedlot are shown in Table 24.

TABLE 24 – CONTRIBUTION TO GLOBAL WARMING POTENTIAL FOR SHORT FED AND LONG FED LOT FED BEEF AT THE FEEDLOT (GATE TO GATE ASSESSMENT)

Supply chain stage / emission source	kg CO_2-e / kg LWT Gain – Supply chain 1 (SF)	Proportion %	kg CO_2-e / kg LWT Gain – Supply chain 2 (LF)	Proportion %
Enteric methane	2.9	39.9	5.26	46.4
Ration production	2.23	29.9	2.84	25.1
Feedpad nitrous oxide and methane	1.5	20.7	2.33	20.6
Remaining processes	0.75	10.0	0.9	7.9

Results presented here are based on an expansion of the system boundary for previous LCA data collection projects (FLOT.328 – Davis & Watts 2006). However, results for the previous study and subsequent research (B.FLT.0339) were presented using a functional unit based on meat yield (HSCW) at the feedlot. This was derived by converting liveweight to HSCW, though obviously the product leaving the feedlot is live cattle. It was felt in the current study that the conversion of liveweight to HSCW introduces a degree of complication to the results and may introduce error in the calculations because HSCW cannot be directly measured at the feedlot.

To address this, results from Davis & Watts (2006) and published LCA results (Peters et al. 2010) have been converted to a LWT basis. This was done in a simple, standard way by multiplying the results by 0.54 (an average slaughter dressing percentage). These results are presented in Table 25.

TABLE 25 – COMPARISON OF GWP FROM AUSTRALIAN FEEDLOT LCA RESEARCH ON A LIVEWEIGHT BASIS – KG CO_2-E / KG LWT GAIN

Reference	MLA project	kg CO_2-e / kg LWT Gain
Davis & Watts 2006	FLOT.328 (2002, 2004)	2.4 – 6.1
Peters et al. 2010	COMP.094 (2004)	4.5[1]
This study	B.FLT.0360	7.5 – 11.3

[1] This number estimated from table 1 in Peters et al. 2010, where CF at the feedlot is reported as 5.5 kg / HSCW. A ratio conversion between the data in figure 1 (mass allocation) and the data in table 3 (unallocated) was used to convert this figure for comparison on an unallocated basis.

Emissions from the present study are higher than those estimated previously. This is partially explained by the following factors:

- This study used a more comprehensive assessment of the feedlot ration than Peters et al. (2010). Upstream commodity burdens were beyond the scope of Davis & Watts (2006) and were not calculated.
- This study used a more comprehensive assessment of manure and effluent emissions than Davis & Watts (2006) or Peters et al. (2010).

To investigate manure emissions further, these were modelled separately for both supply chains.

8.1.1 Manure management emissions

Estimation of manure and effluent emissions at the feedlot were based on a theoretical mass balance (explained in 7.2.2). This included some expert judgements about emission sources where research gaps exist. Manure emissions are presented using the same functional unit as the gate-to-gate assessment (CO_2-e per kilogram of LWT gain) and represent a sub-set of these emissions.

8.1.1.1 Theoretical mass balance

Results from the theoretical mass balance show emissions of 1.98 kg CO_2-e / kg LWT gain for short fed cattle and 3.09 kg CO_2-e / kg LWT gain for long fed cattle. Emission sources for each supply chain are presented in Figure 13 and Figure 14, showing all emissions sources. Emission sources indicate both the source (i.e. feedpad) and the emission type (i.e. methane – CH_4 or nitrous oxide – N_2O).

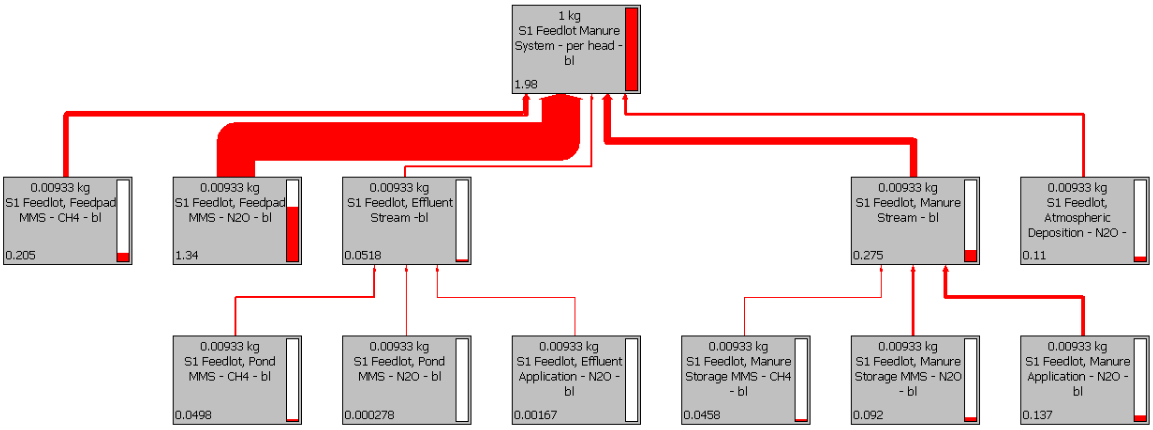

FIGURE 13 – GLOBAL WARMING POTENTIAL (GWP) OF MANURE EMISSIONS FOR DOMESTIC SHORT FED BEEF CATTLE PRODUCTION AT THE FEEDLOT (SUPPLY CHAIN 1 – PER KG LWT GAIN)

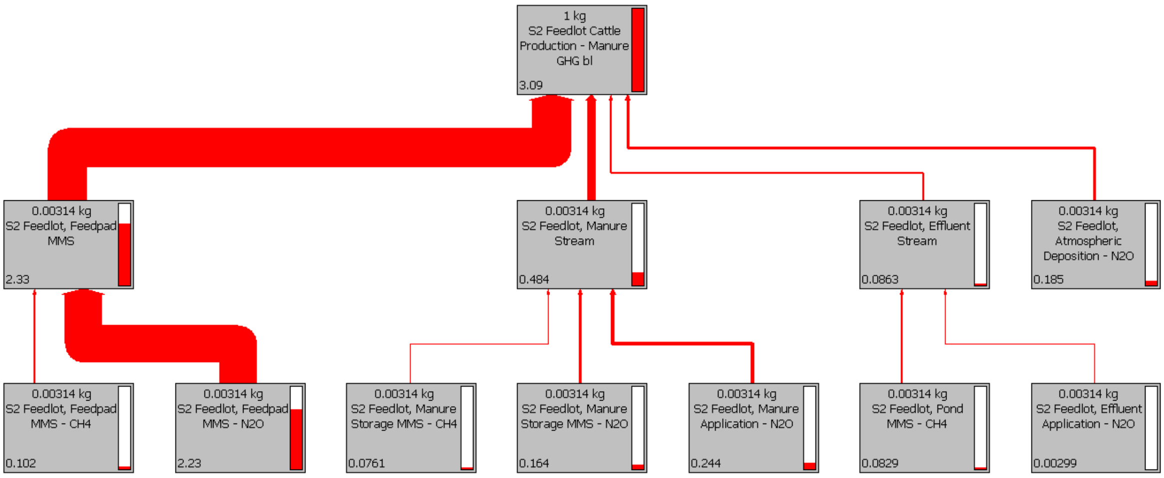

FIGURE 14 – GLOBAL WARMING POTENTIAL (GWP) OF MANURE EMISSIONS FOR EXPORT LONG FED BEEF CATTLE PRODUCTION AT THE FEEDLOT (SUPPLY CHAIN 2 – PER KG LWT GAIN)

The theoretical mass balance model included all emission sources that are defined by the current DCC methodology (feedpad nitrous oxide and methane, and nitrous oxide emissions from atmospheric deposition of ammonia and manure application emissions) together with additional emission sources identified in the literature.

The theoretical mass balance selectively used factors defined by the DCC (2007a) for Australian feedlots and other values found in the literature, based on expert judgement. Factors that were taken directly from the DCC (2007a) method are as follows:

- Nitrous oxide emission factor from the feedlot pad / storage (2% of excreted N).
- Methane emission factor from the feed pad.
- Nitrous oxide emission factor for manure application.

Factors that vary from the DCC (2007a) method are as follows:

- Ammonia volatilisation factor. The theoretical mass balance follows the NPI ammonia emission calculations (total ammonia-N = 82%) compared to the DCC emission factor of 30%.
- The emission factor for nitrous oxide from atmospheric deposition. A factor of 0.2% of volatilised ammonia was used (lower limit recommended by the IPCC – Dong et al. 2006). The DCC recommend a default value of 1% based on the IPCC default, from research done primarily in northern hemisphere countries. This is considered too high when compared to the conditions experienced in Australia and the findings of other research, such as the emissions of nitrous oxide from N fertilisers which are considerably lower than IPCC defaults.

These assumptions are yet to be substantiated by research, but are provided here as an indication of likely scenarios. To enable comparison with the standard DCC method, a separate scenario was modelled.

8.1.1.2 DCC scenario

For comparison, manure emissions from the short fed feedlot were modelled using the emission factors and sources outlined by the DCC (2007a). The DCC identify fewer emission sources than covered by the theoretical mass balance, and prescribe some different emission factors compared to those used in the theoretical mass balance (as described).

Overall, emissions using the DCC method for manure emissions during lot feeding (short fed) were (2.15 kg CO_2-e / kg LWT gain), or 9% higher than the theoretical mass balance approach. Emission sources are shown in Figure 15.

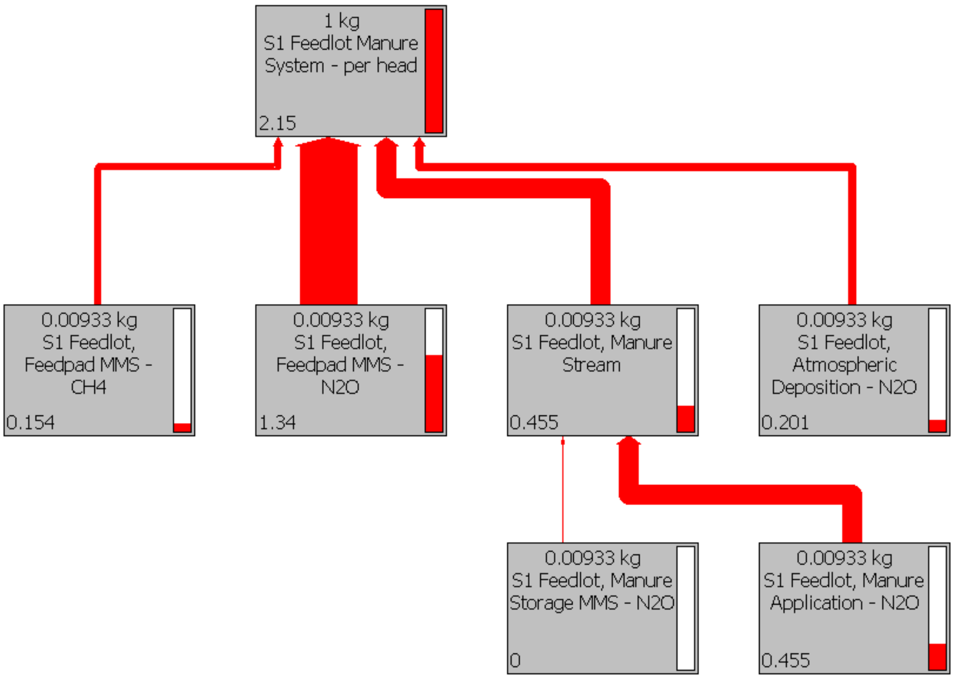

FIGURE 15 – GLOBAL WARMING POTENTIAL (GWP) OF MANURE EMISSIONS FOR DOMESTIC SHORT FED BEEF CATTLE PRODUCTION AT THE FEEDLOT USING THE DCC METHOD (SUPPLY CHAIN 1 – PER KG LWT GAIN)

The DCC identify five emission sources related to feedlot manure management (Figure 15). A single emission factor is supplied for nitrous oxide from 'solid storage and drylot' which has been simplified here as an emission from the feedpad only. Two emission sources (feedpad and storage nitrous oxide, and feedpad methane) are also significant contributors to the theoretical mass balance emission model.

8.2 Primary energy usage

PE includes both direct and indirect energy usage associated with beef production. A whole of supply chain assessment is provided for context, which includes energy usage associated with upstream (cattle breeding and backgrounding) and downstream (meat processing) processes.

At the feedlot, the energy usage assessment has been based on data collected by Davis et al. (2008), with the following additions:

- Upstream energy usage data have been modelled that relate to the supply of energy, associated with the energy industry. This includes 'upstream uses' such as line losses for electricity and transportation of diesel.
- Upstream energy use is associated with the production of grain at the farm, where energy is used for tillage and harvesting, and is 'embedded' within fertiliser.

PE usage ranged from 34.5 to 49.1 MJ / kg LWT gain for the supply chain 1 (short fed) and supply chain 2 (long fed) respectively. Energy embedded within feed contributed 89 – 90% of total energy at the feedlot, which included minor contributions from transport and fuel use at the feedlot for feed delivery. Contributions to energy usage for supply chain 2 are shown in Figure 22 (Appendix 1).

Energy usage was dominated by upstream grain production. Energy intensity with grain production is dominated by the production of urea (see Figure 22) which is used broadly by Australian farmers. Some feedlots may have an opportunity here by growing 'low energy embedded grain' on their farms with the use of feedlot manure as an alternative to urea. This could reduce the energy burden of feedlot cattle to some degree. However, it should be noted that feedlot 2 (long fed) does apply a proportion of the manure produced in the feedlot to produce grain, hay and silage which is fed back in the ration. This was taken into account in the assessment. None-the-less, embedded energy in feed at this feedlot was still high.

To compare the values estimated in this study with previous research, energy usage from Davis & Watts (2006) has been back-calculated to present results on a liveweight gain basis (Table 26).

TABLE 26 – COMPARISON OF PRIMARY ENERGY USAGE AT AUSTRALIAN FEEDLOTS ON A LIVEWEIGHT BASIS MJ / KG LWT GAIN

Energy use	Davis & Watts (2006)	Supply chain 1 - SF	Supply chain 2 - LF
Direct usage at the feedlot	0.4 – 4.9	3.4	5.4
Indirect usage (transport, commodities where applicable)	0.2 – 4.7	31.1	43.7
Total	**0.6-9.6**	**34.5**	**49.1**

[1,] This breakdown will slightly underestimate direct usage because feed delivery has been grouped under feed ration (labelled an indirect use here). [2] Indirect energy usage will be slightly elevated for the reason given above.

8.3 Water usage

Blue water usage at the feedlot is based on data from Davis et al. (2008) for two feedlots, with additional water usage associated with irrigated feed inputs and water capture on-site as effluent. Water usage at feedlot 1 (short fed) was 871 L / kg LWT gain, while water usage at feedlot 2 (long fed) was 151 L / kg LWT gain. The breakdown of blue water usage is shown in Figure 16 and Figure 17.

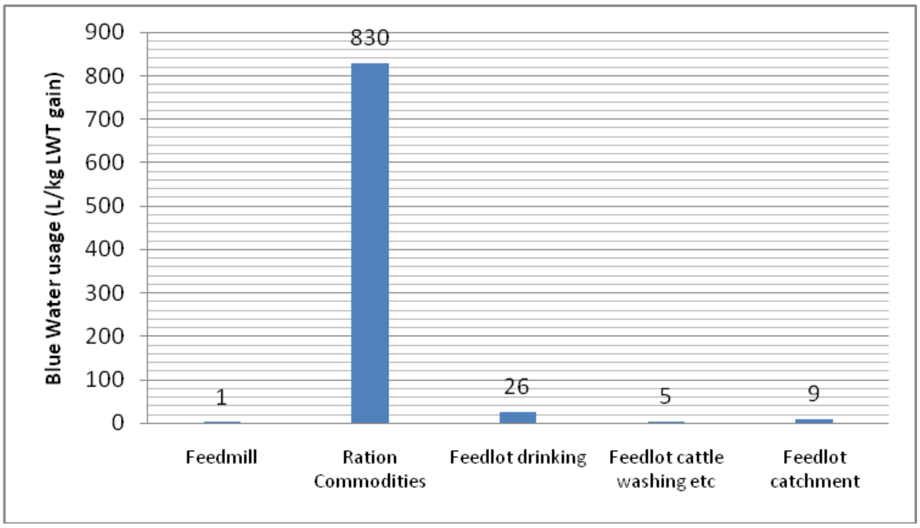

FIGURE 16 – BLUE WATER USAGE FOR SUPPLY CHAIN 1 (SHORT FED) PER KG OF LWT GAIN

Figure 16 shows the dominating effect of irrigated feed on feedlot water usage (as discussed in section 9.2.3). It is not clear what proportion of the feedlot industry relies on commodities produced from irrigated inputs, but it is worth noting that some common commodities such as cotton seed may have a significant water burden (up to 1000 L / kg). The impact of this requires further review across the feedlot sector.

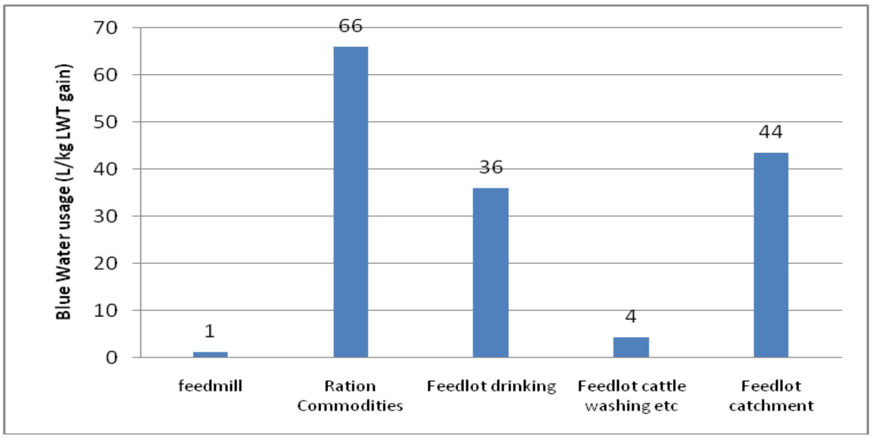

FIGURE 17 – BLUE WATER USAGE FOR SUPPLY CHAIN 2 (LONG FED) PER KG OF LWT GAIN

Figure 17 shows the blue water usage for feedlot 2 (long fed) is considerably lower than water usage at feedlot 1 (short fed), mainly as a result of the lower water burden with commodities. However, when the water associated with commodities was removed, this feedlot used double the water of feedlot 1 – short fed (Table 27). Green water usage associated with growing commodities is also shown in Table 27.

TABLE 27 – BLUE AND GREEN WATER USAGE FOR TWO AUSTRALIAN BEEF FEEDLOTS (L / KG LWT GAIN)

Stage in supply chain	Water type	Supply Chain 1 (short fed)	Supply Chain 2 (long fed)
Direct feedlot water usage	Blue water	41	85
Ration commodities irrigation		830	66
Ration commodities rainwater	Green water	1,175	2,974

For comparison, data from Davis et al. (2008) is shown in Table 28 with results presented on a LWT basis.

TABLE 28 – WATER CONSUMPTION IN FEEDLOT ACTIVITIES (DAVIS ET AL. 2008)

Major Areas of Water Consumption	Water Usage L/kg LWT gain	Percent of Total Water Consumption
Drinking Water	12 – 47	78 – 91%
Feed Processing	0.2 – 1.3	1 – 6%
Cattle Washing	3 – 6	0 – 12%
Administration	0.3 – 1.8	0 – 5%
Sundry Uses	0.2 – 8	0 – 7%

Direct water usage for feedlot activities reported in Table 28 are similar to those found by this study (both studies are based on the same dataset).

9 Discussion

9.1 GHG emissions from manure management

A specific aim of this study was to investigate feedlot manure system emissions in detail. Results for the two supply chains, together with the DCC scenario are presented in Table 29 along with manure emissions previously reported by Davis & Watts (2006). In order to compare with this reference, the Davis & Watts results were back-calculated to present values on a liveweight gain basis.

TABLE 29 – COMPARISON OF GWP FROM MANURE MANAGEMENT SYSTEMS AT AUSTRALIAN FEEDLOTS ON AN UNALLOCATED BASIS – KG CO_2-E / KG LWT GAIN

Emission source	Davis & Watts (2006)	Theoretical mass balance SF	Theoretical mass balance LF	DCC (2007a) methodology SF
Storage and Feedpad N_2O	0.65-1.62	1.34	2.23	1.34
Feedpad CH_4	0.02-0.14	0.21	0.10	0.154
Manure Storage (N_2O)		0.09	0.16	
Manure Storage (CH_4)		0.05	0.08	
Effluent Pond (N_2O)		0.000	0.003	
Effluent Pond (CH_4)		0.05	0.08	
Manure Application (N_2O)		0.14	0.24	0.455
Effluent Application (N_2O)		0.002	0.003	
Atmospheric deposition (N_2O)		0.11	0.19	0.201
Totals	**0.67-1.76**	**1.98**	**3.09**	**2.15**

Table 29 shows the more comprehensive assessment of emissions for the two scenarios modelled in this study, resulting in a trend towards higher manure emissions. Manure application and atmospheric deposition were not included by Davis & Watts (2006) as these were outside the system boundary defined for the project. It is not clear if these emissions were included by Peters et al. (2010).

When comparing the theoretical mass balance and the DCC scenario for the short fed system, the DCC scenario resulted in the highest emissions because of the elevated emissions associated with manure application, and elevated atmospheric deposition values. These emissions differ because:

- The DCC use a lower rate of N volatilisation from the feedlot, resulting in **more nitrogen being predicted for land application** (and subsequent losses) and **less nitrogen being predicted for atmospheric deposition** and re-emission as nitrous oxide.
- Emissions from atmospheric deposition still remain higher for the DCC scenario because the theoretical mass balance applied a lower emission factor.

These two emission sources are particularly sensitive to the mass balance of nitrogen at the feedlot, particularly the loss of nitrogen as ammonia. The DCC (2007a) suggest that volatilisation of nitrogen only occurs at a rate of 30% from the whole feedlot, compared with 85% losses estimated by the NPI (FSA Consulting 2006). However, the lower rate of N loss estimated by the DCC is not considered plausible because this would result in manure nitrogen concentrations in the order of 15%.

Hence, recommending changes to the DCC ammonia volatilisation factors would decrease the estimated amount of nitrogen available for loss at the point of manure application (reducing nitrous oxide emissions at this point) and increase atmospheric deposition losses unless combined with a lower emission factor for atmospheric deposition.

A summary of proportional emissions from the theoretical mass balance and the DCC scenarios is provided in Table 30.

TABLE 30 – PROPORTIONAL GHG EMISSIONS FROM MANURE MANAGEMENT SYSTEMS USING TWO APPROACHES – THEORETICAL MASS BALANCE AND THE DCC METHODOLOGY

Emission source	Theoretical mass balance (av)	DCC (2007a) methodology SF
Storage and Feedpad N_2O	68-72%	62%
Feedpad CH_4	3-10%	7%
Manure Storage (N_2O)	5%	-
Manure Storage (CH_4)	2%	-
Effluent Pond (N_2O)	0.01 – 0.1%	-
Effluent Pond (CH_4)	3%	-
Manure Application (N_2O)	7-8%	21%
Effluent Application (N_2O)	0.1%	
Atmospheric deposition (N_2O)	6%	9%

Table 30 shows that storage and feedpad nitrous oxide emissions dominate manure emissions at the feedlot. Interestingly, the next highest emission source when following the DCC method is nitrous oxide from manure application. Nitrous oxide from atmospheric deposition is also relatively high following the DCC methodology. Both of these emissions are sensitive to the DCC ammonia volatilisation factor, which is considered to be too low.

Carbon sequestration from carbon in manure has also been raised as an area of potential emission reduction for feedlots (Redding 2010). To investigate the potential impacts of sequestration, a scenario was modelled that included carbon sequestration for the manure system at the short fed feedlot (Figure 18). The sequestration rate was estimated to be equal to 13 kg of CO_2-e / finished animal, or approximately 10% of the carbon available for land application.

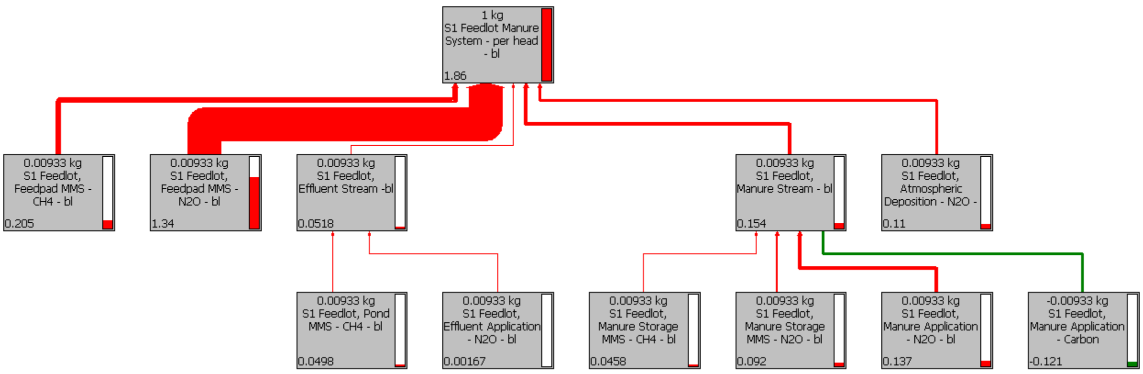

FIGURE 18 – GLOBAL WARMING POTENTIAL (GWP) OF MANURE EMISSIONS FOR DOMESTIC SHORT FED BEEF PRODUCTION AT THE FEEDLOT INCLUDING CARBON SEQUESTRATION (SUPPLY CHAIN 1 – PER KG LWT GAIN)

Including carbon sequestration reduced overall manure GHG emissions by 6% at this rate of sequestration. It is noted that sequestration rates can vary greatly (from 3 to 50%), however higher sequestration rates were associated with higher application rates than are commonly used in Australia.

9.1.1 Manure emission sensitivity

A sensitivity check of the main emission factors for manure management showed the model to be sensitive to changes in the pad nitrous oxide emission factor, ammonia volatilisation factor and the manure application emission factor. The sensitivity analysis was done using upper and lower emission factor values provided by the IPCC (Dong et al. 2006; De Klein et al. 2006).

The IPCC (Dong et al. 2006) identifies a range of nitrous oxide emissions from feed pads of 1-4%, with a default value of 2% (which is recommended for use in Australia by the DCC (2007a)). The sensitivity of the model to changes in this factor is shown in Figure 19.

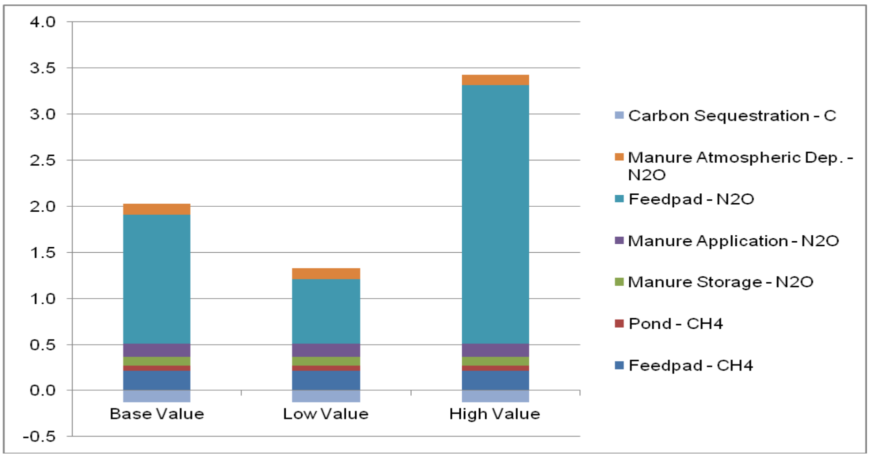

FIGURE 19 – SENSITIVITY OF MANURE GWP TO FEEDPAD NITROUS OXIDE EMISSIONS (SUPPLY CHAIN 1 – PER KG LWT GAIN)

Changes in the pad nitrous oxide emission factor will alter manure emissions by -35% to +69%. For the feedlot sector (gate-to-gate), changes to pad nitrous oxide emissions result in a -9.3 to +18.6% change in overall emissions. The sensitivity of this factor is even greater when factors not controlled by the feedlot (such as upstream grain production) are removed.

9.2 Supply chain context

9.2.1 Global warming potential

To contextualise results from the feedlot, an extended supply chain system was constructed which included breeding, backgrounding, and slaughter of cattle in addition to the feedlot component. Data for these operations were based on simplified data collected in previous research and literature (Peters et al. 2009a, MLA 2002), expert knowledge and some foreground data collection from a Queensland meat processor. The same breeder supply chain was used for both feedlots, so that differences in the overall result are mainly in response to the feedlot component of the supply chain. One variation to this was the inclusion of a backgrounding component in the long-fed supply chain to grow steers from 360 kg (entry weight for the short fed feedlot) to 440 kg (entry weight for the long fed feedlot). Importantly, the breeding supply chain was a Queensland system which has higher enteric methane emission factors compared to the southern states in the DCC methodology.

To enable the contribution of the feedlot to be compared with the breeding sector, the emissions for both breeding, backgrounding (for the long fed supply chain) and for the feedlot were disaggregated up to immediately prior to slaughter. This was done to avoid complications related to allocation processes at the meat processing plant. These results are presented in Table 31 and Table 32.

TABLE 31 – CONTRIBUTION OF SUPPLY CHAIN STAGES TO GWP (PRESENTED AS KILOGRAMS OF LIVEWEIGHT PRIOR TO SLAUGHTER – SHORT FED)

Supply chain stage	kg of LWT gained	Proportion of final liveweight (%)	kg CO_2-e / kg LWT gain	Total CO_2-e emissions
Breeding – through to yearling	361	77	13.7	4946
Feedlot	107	23	7.6	813
Total	468	100	12.3 [1]	5759

[1] This value is the weighted average over the whole life of the animal

TABLE 32 – CONTRIBUTION OF SUPPLY CHAIN STAGES TO GWP (PRESENTED AS KILOGRAMS OF LIVEWEIGHT PRIOR TO SLAUGHTER – LONG FED)

Supply chain stage	kg of LWT gained	Proportion of final liveweight (%)	kg CO_2-e / kg LWT gain	Total CO_2-e emissions
Breeding – through to yearling	361	48	13.7	4946
Backgrounding	80	11	9.2	736
Feedlot	319	42	11.3	3605
Total	760	100	12.2 [1]	9286

[1] This value is the weighted average over the whole life of the animal

Table 31 and Table 32 are presented to allow comparison of production systems prior to slaughter, and are presented on like terms to the feedlot results (kg LWT gain and final kg LWT produced prior to slaughter). Considering they exclude emissions associated with meat processing and have not undergone any allocation process, *they are not easily re-calculated to a HSCW basis*.

Table 31 and Table 32 show the large GWP burden associated with breeding and the increasing efficiency of production as the young cattle grow. These results are in agreement with Charmley et al. (2008) and Hunter & Niethe (2009), though both of these studies investigated enteric methane emissions only. Compared to these studies, the comparative advantage of lot feeding is not as great when other emissions (embedded emissions with grain, nitrous oxide etc) are included. This being said, a comparison between the short fed feedlot (GWP of 7.5 kg CO_2-e / kg LWT gain) and the backgrounding stage of the long fed supply chain (GWP of 9.2 kg CO_2-e / kg LWT gain) shows that cattle on high performance diets perform favourably compared to backgrounding on grass. It is noted again that this is a fairly crude comparison and needs to be subjected to more rigorous investigation however. To create a fairer comparison the cattle need to be grown to a similar slaughter weight and the pasture / forage crop system needs to represent regular practices and cattle performance.

There is a natural reduction in GWP intensity for slaughter cattle as the animals grow, because the additional liveweight dilutes the contribution of the breeding herd. This is balanced by the efficiency of liveweight gain to methane emissions, and the effect is strongest when highly efficient systems, such as lot feeding or rapid growth rates on grass are achieved.

Interestingly, the results for each supply chain were very similar despite the differences in contributions from each stage of the supply chain and the different emission intensities of the feedlot stage. While it may be expected that the domestic (short fed) supply chain would be superior based on the feedlot results, it must considered that the short period of time on feed means that the feedlot is only partially able to offset the breeding burden. The long fed supply chain on the other hand is less efficient at the feedlot stage, but is growing the cattle to a greater final weight, which more effectively dilutes the breeding burden. It would follow that a longer period of high efficiency feeding (such as 140-180 days on feed) would be the most efficient system with respect to GWP.

These results were also taken through to post slaughter to allow comparison with the literature. This was done using the functional unit '1 kg of HSCW'. Whole of supply chain emissions for the two supply chains were 16 kg CO_2-e / kg HSCW for both domestic (short fed) and export (long fed) beef. The results are not different because all factors at the meat processing facility were kept equal per kilogram of meat processed. With respect to primary emission sources at the supply chain level, enteric methane from the breeding herd dominated overall emissions for both systems, contributing in the order of 51-82% of GWP for the long fed and short fed systems respectively. Feedlot manure emissions (feedpad and stockpile) contributed 7.9% and 2.8% of overall emissions for the long fed and short fed systems respectively. The greater contribution for the long fed system is a function of the proportion of liveweight gain occurring in the feedlot (see Table 32).

These results show a higher proportion of the GWP burden being associated with enteric methane than most other researchers (76 – 87% for this study compared to 43 – 69% for three studies reported in the literature (Barber et al. 2007; Nemry et al. 2001; Verge et al. 2008 – see Table 6). This may be partly in response to the streamlined approach taken in this scoping study, which may have omitted some upstream processes and associated emission sources.

Results for the whole supply chain tend to be higher than other estimates in the literature, and higher than those provided by Peters et al. (2010) for the supply chain which included a feedlot (NSW). For comparison with other studies, a truncated version of Table 6 is presented (Table 33) with the results from this study presented on like terms (un-allocated), meaning that *all emissions are transferred to the carcass weight without allocating any burdens to other by-products*. This is the least favourable allocation method possible, but is useful for the sake of comparison.

TABLE 33 – GLOBAL WARMING POTENTIAL OF BEEF PRODUCTION FROM A RANGE OF LITERATURE SOURCES

Reference	Country	System	Results on standard basis kg CO_2e/kg HSCW – unallocated[1]
Nemry et al. (2001)	Belgium	-	14.8
Peters et al. (2010)	AUS (VIC 2004)	Organic	18.1
	AUS (NSW 2002)	pasture/feedlot	15
	AUS (NSW 2004)	pasture/feedlot	15.4
Vergé et al. (2008)	Canada	pasture / Feedlot	19.2
This study	**Supply chain 2**	**pasture / feedlot export long fed**	**22.2**
	Supply chain 1	**pasture / feedlot domestic short fed**	**22.4**
Weidema et al. (2008b)	EU-27	Feedlot/pasture	28.7
Cederberg et al. (2009)	Brazil	Pasture	28.0
Ogino et al. (2004)	Japan	Long-fed feedlot	32.3

In comparing these results to previous Australian research (Peters et al. 2010) the higher emissions are partially explained by:

- Higher GHG emissions from livestock at the breeder farm for this study. Following the DCC (2007a) methodology, Queensland grazing cattle have a higher methane emission rate compared to southern NSW and Victorian supply chains.
- Higher emissions from the feedlot component modelled in this project compared to the feedlot modelled by Peters et al. (2010).
- More comprehensive assessment of GHG emissions associated with commodity production at the feedlot for this study compared to Peters et al. (2010).

It is also noted that this scoping study did not include every emission source at every point and used optimistic herd parameters for the upstream supply chain (calving rate of 85% and growth rates of around 0.9 kg / day for slaughter cattle). Consequently, overall emissions are likely to be higher still as more realistic and comprehensive data are collected in the Queensland supply chain LCA being conducted currently through MLA (B.CCH.2028). This being said, the full supply chain LCA will incorporate carbon offsets associated with on-farm sequestration during grazing, which this scoping study has not included.

9.2.2 Primary energy

PE usage for supply chain 1 (short fed) totalled 14.3 MJ / kg HSCW, while PE for supply chain 2 (long fed) was 32.2 MJ / kg HSCW (higher heating value – HHV). Energy usage for the whole supply chain was dominated by energy used in commodity production for lot feeding, which contributed approximately 71% of total energy usage. Total energy usage figures are of a similar order to Peters et al. (2010) and other LCA studies of beef production.

9.2.3 Water usage

Water usage for feedlot beef production in this study was higher than previously estimated by Peters et al. (2009a). This is explained largely by differences in the methodology between the two studies, and by the more comprehensive assessment of embedded irrigation water in this study.

Water usage data are presented for 'blue' and 'green' water. Blue water usage for supply chain 1 (short fed) is shown in Figure 20 on a 'L/kg HSCW' basis. Total blue water usage for supply chain 2 (long fed) is shown in Figure 21. Blue and green water usage is summarised for both supply chains in Table 34.

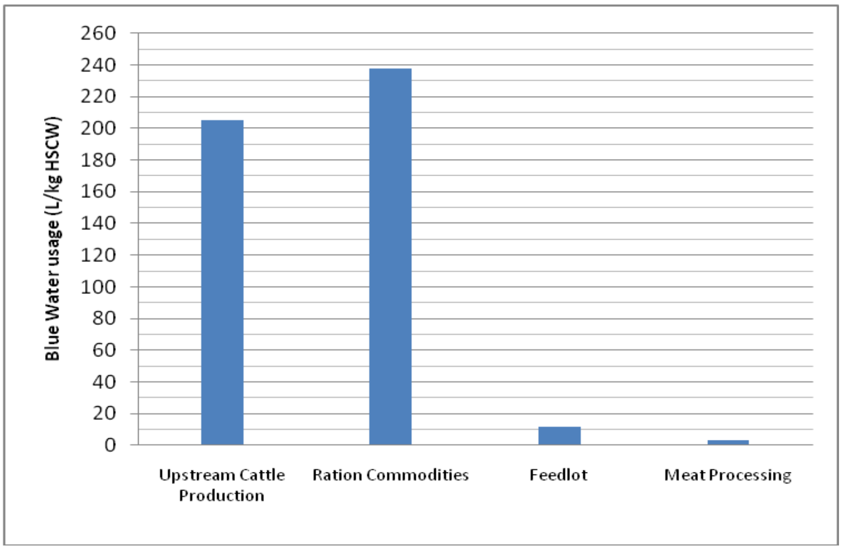

FIGURE 20 – BLUE WATER USAGE FOR SUPPLY CHAIN 1 (SHORT FED) PER KG OF HSCW

Blue water usage for supply chain 1 was dominated by water used in upstream cattle breeding (drinking water) and water used for the production of irrigated ration commodities.

Drinking water was estimated for each livestock class over one year from average water requirements. Irrigation water used with ration commodities was estimated for two commodities, silage and cotton seed. Silage irrigation is practiced at many feedlots and was calculated at a fairly conservative rate of 4 ML/ha for a 20.5 t / ha (dry matter) corn silage crop.

Irrigation water usage associated with cotton seed production was estimated from an economic allocation of water use to cotton seed and lint, based on average irrigation requirements and yields of cotton in Australia. This resulted in 1 ML of irrigation water used / tonne of cotton seed. These results are based on a rapid assessment and would warrant further research, considering their importance to the overall blue water use for beef production. The overall contribution of water used directly by the feedlot was small, which is not surprising considering the short feeding period.

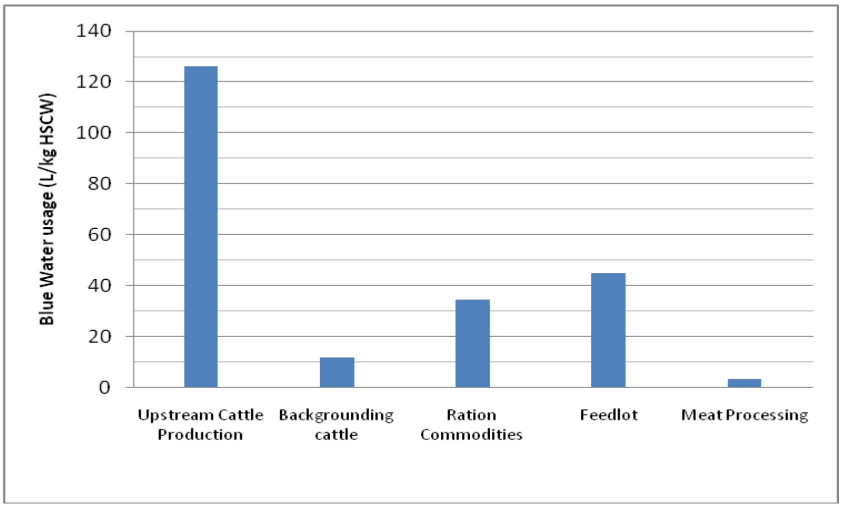

FIGURE 21 – BLUE WATER USAGE FOR SUPPLY CHAIN 2 (LONG FED) PER KG OF HSCW

Blue water usage for supply chain 2 (Figure 21) also showed that the feedlot makes a small contribution to water usage throughout the supply chain. Water usage is dominated by upstream cattle breeding (water use for drinking). Ration commodities make up a smaller proportion of water usage at this feedlot because of the lower use of on irrigated commodities in the ration.

Cattle production also 'uses' a considerable amount of water derived from rainfall for pasture and grain production. This 'green' water is categorised separately as it does not have the same degree of transferability or impact on other uses that blue water may have. To enable comparison with other literature, blue and green water usage data are presented in Table 34.

TABLE 34 – BLUE AND GREEN WATER USAGE FOR TWO AUSTRALIAN BEEF PRODUCTION SUPPLY CHAINS (PER KILOGRAM OF HSCW)

Stage in supply chain	Water type	Supply Chain 1 (short fed)	Supply Chain 2 (long fed)
Total Blue Water	**Blue water**	**460**	**222**
Ration Commodities	Green water	1,469	3,720
Pasture production (upstream)	Green water	20,582	14,358
Total Green Water	**Green water**	**21,146**	**18,390**

Total water usage (blue water + green water) is roughly comparable to a water footprint or virtual water assessment of beef, as presented in the literature. Literature estimates have been re-presented here for comparison as L / kg HSCW (Table 35).

TABLE 35 – VIRTUAL WATER USE ESTIMATES FOR BEEF AND OTHER MEAT PRODUCTS (L / KG HSCW)

Species	L / kg HSCW (Australian estimates)	L / kg HSCW (World average estimates)	Reference
Beef	17,112	15,497	Hoekstra & Chapagain 2007
Beef (pasture / feedlot)	**18,612 – 21,606**	**-**	**This study**
Sheep meat	6,947	6,143	Hoekstra & Chapagain 2007
Pork	2,753 – 3,020	-	Wiedemann et al. 2010b
Pork	5,909	4,856	Hoekstra & Chapagain 2007
Chicken meat	2,914	3,918	Hoekstra & Chapagain 2007

In an attempt to assess the likely industry wide blue water usage for beef production, Wiedemann et al. (2010a) compared water use for irrigated feed, pasture and feedlot feed (from ABS data) with national beef production. This provides an indication of likely water usage across the beef industry (Table 36).

TABLE 36 – WATER USE CONTRIBUTION FOR BEEF PRODUCTION FROM IRRIGATED PASTURES, CROPS AND DRINKING WATER USING ABS DATA

Water source	Australian water use (ABS 2008)	"Best guess" water allocation to the beef industry	Water (ML) allocated to the beef industry	L Water per kg HSCW beef
	ML/yr	%	ML/yr	L / kg beef*
Irrigated cotton	867,662	5	43,383	20
Cereal crops for grain / seed	674,470	10	67,447	31
Irrigated pasture (inc. lucerne) for hay / silage	794,622	20	158,924	74
Irrigated pasture for meat cattle grazing	512,874	100	512,874	238
Cereal crops for hay / grazing	150,984	40	60,394	28
Uses other than irrigation	885,234	20	177,047	82
Totals			**1,020,069**	**474**

* Water usage divided by national beef slaughter (2,151,237 tonnes) from ABS statistics for 2006.

The water usage data presented here are comparable to the total blue water usage estimates from this study.

It should be noted that this was done at the national level, and the allocation of 'total water to total beef' will not be representative at the farm level. However, it provides an indicative comparison and suggests that further research into water usage for beef production is warranted to gain more reliable estimates than have been generated to date.

10 Conclusions and recommendations

10.1 Conclusions

This scoping study investigated two supply chains, with attention being focussed on two feedlots within the supply chain. Both supply chains used generic upstream and downstream processes (such as cattle breeding and meat processing) as much as possible, to highlight the differences between the lot-feeding systems.

As with all LCA studies, allocation processes can greatly influence the final results. This study used a ‘middle of the road’ allocation technique, where burdens were equally applied to all slaughter by-products of value, but not direct wastes. This is more favourable than the ‘unallocated’ values presented by Davis & Watts (2006) and Davis et al. (2008) and slightly less favourable than the mass allocation process applied by Peters et al. (2009a, 2010).

The study focused on two very different feedlots (one smaller feedlot feeding for domestic markets and one larger feedlot feeding for long-fed export markets). These two feedlots could not be considered representative of the whole industry. As the results are preliminary in nature, these should not be considered as industry averages without further research.

10.1.1 Global warming potential

GWP at the feedlot was 7.5 kg CO_2-e / kg LWT gain for supply chain 1 (short fed) and 11.3 kg CO_2-e / kg LWT gain for supply chain 2 (long fed). The major contributions to emissions were enteric methane (about 40 – 45%), ration production (about 25 – 30%) and feedpad emissions (about 20%). All other contributions (energy usage, manure storage, treatment and reuse etc) amounted to approximately 10%. The higher emissions for the long fed supply chain are primarily driven by the lower production efficiency (feed conversion) of very long fed cattle (> 300 days) compared to short fed cattle (70 days). It is noted that assessment of feed grains and other commodities relied on a series of desktop studies. Considering their importance at the feedlot, further research in this area is warranted to ensure correct data are used.

From an analysis of the manure management system at the feedlots the results from the theoretical mass balance show emissions of 1.98 kg CO_2-e / kg LWT gain for short fed cattle and 3.09 kg CO_2-e / kg LWT gain for long fed cattle. An analysis was also conducted for the short fed scenario using the DCC methodology and manure management system emissions contributed 2.15 kg CO_2-e / kg LWT gain, or 8.6 % higher than the theoretical mass balance approach.

This study provided a more comprehensive assessment of manure GHG than previous feedlot research such as Davis & Watts (2006). However, the accuracy of the findings are limited by the available research into nitrogen and volatile solids flows at feedlots, and specific emissions factors. To date, there are large gaps in the knowledge of these issues for Australian feedlots.

Assessment of GHG emissions at feedlots can be divided into a regulatory approach and a theoretical approach. Emissions are estimated as part of the national inventory using methods developed by the DCC (2007a), which will form the basis for any regulation of GHG in the future. This approach has been found to be simplistic in nature, and relies on emission factors that have not been validated in Australian conditions.

The theoretical mass balance approach is an alternative, more comprehensive and representative framework for assessing feedlot GHG. However, it is also limited by the factors available for estimating emissions.

This project has identified the key emission factors shared by both approaches, allowing prioritisation of R&D needs for the industry. These are summarised in Table 37, based on the proportional emission sources identified in Table 30.

TABLE 37 – KEY EMISSION FACTORS AND RESEARCH NEEDS FOR ESTIMATING MANURE GHG

Emission source	Assessment method	Contribution to Manure GHG	R&D Ranking
Storage and Feedpad N_2O	Mass balance / DCC	62 – 72%	1
Atmospheric deposition (N_2O)	DCC	9%	2
Ammonia volatilisation (NH_3)	Mass balance	indirect	2
Feedpad CH_4	Mass balance / DCC	3 – 10%	3*
Manure Application (N_2O)	Mass balance / DCC	21%	4
Manure Storage (CH_4)	Mass balance	2%	
Effluent Pond (N_2O)	Mass balance	0.01 – 0.1%	
Effluent Pond (CH_4)	Mass balance	3%	
Effluent Application (N_2O)	Mass balance	0.1%	

* Because of the interrelationship between factors, research areas may need to be grouped.

The interrelationship between ammonia emissions and nitrous oxide from manure application and atmospheric deposition highlight the importance of a mass balance approach to research in this area. Mass balance theory is essential to the DCC method of approach (as emission factors are all related back to nitrogen intake and excretion. Considering this, research in this area needs to use an integrated mass balance approach ensure accurate results and emission factors are generated. Although not studied in this LCA, different nitrogen intake levels will also influence manure GHG, possibly to a large extent.

It should be noted that three relevant emission factors provided by the DCC are much higher than would be reasonably assumed from the literature and mass balance theory. These are:

- Ammonia emissions. Applying a mass balance theory to feedlot nitrogen shows that, with the emission factor of 30% supplied by the DCC, manure nitrogen levels are in the order of 15% (on a DM basis). Documented manure analyses from Australian feedlots average 2.2% N, suggesting the loss rate of nitrogen from the feedlot is significantly higher. This is supported by the NPI (FSA Consulting 2006). The consequences of this are that manure application emissions from nitrous oxide are far higher than is likely in reality. Changing the ammonia emission factor will lead to higher losses from atmospheric deposition however, and should be done only in conjunction with emission research from this source.
- Default emission factors for manure applications are higher than would appear reasonable. The IPCC (Dong et al. 2006) do not differentiate between emission factors for organic and inorganic nitrogen applications, but this has not been carried over to the Australian methodology. This is despite the fact that the Australian emission factor (nitrous oxide) from fertiliser nitrogen is 0.3%, while the emission factor from manure is 1% (a threefold difference). Considering this, it may be possible to change the manure application emission factor with a smaller research project because of the supporting literature for fertiliser nitrogen.
- Default emission factors for nitrous oxide from atmospheric deposition are higher than would appear reasonable, considering the low likely deposition rates and low emission rates from Australian soils in general. Considering the supporting literature for other nitrogen sources it may be possible to change this emission factor with a smaller research project.

When the full supply chain was considered for context, GWP was estimated to be 16 kg CO_2-e / kg HSCW for both the domestic short fed and export long fed supply chains. When the results were presented on an unallocated basis, this equates to 22.4 and 22.2 kg CO_2-e / kg HSCW for the domestic short fed and export long fed supply chains respectively. These results are in the range but at the higher end of comparable international studies, and are close to 50% higher than the results reported by Peters et al. (2010). This is due to several cumulative factors, including the use of a Queensland breeding system which has higher enteric methane emissions (calculated with the DCC methodology, compared to the southern states), higher feedlot emissions, more inclusive commodity production emissions at the feedlot and more inclusive modelling of GHG emissions from livestock at the breeder farm.

Results were presented that showed the contribution of the feedlot to reducing overall emissions over the life of the animal. While promising, a comparable grazing system is not shown here and the results should not be taken as a direct confirmation of the efficiency of lot feeding. This is because there is a natural trend towards improved GWP performance as the slaughter animal grows, which occurs on grass or grain provided the growth rates are reasonable for the grass fed system. This will be further explored in the Queensland beef LCA (B.CCH.2028).

It should be noted that this was a scoping study, and did not include some minor emission sources. Consequently total GHG burdens may be higher than this. However, sequestration options and other offsets (such as substitution of manure for fertiliser) will reduce overall emissions. These options will be studied in more detail in further LCA research.

10.1.2 Primary energy usage

PE usage at the feedlot ranged from 34.5 to 49.1 MJ / kg LWT gain for the supply chain 1 (short fed) and supply chain 2 (long fed) respectively. Energy embedded within feed contributed 89 – 90% of total energy at the feedlot, which included minor contributions from transport and fuel use at the feedlot for feed delivery. Energy usage was considerably higher than previously estimated by Davis & Watts (2006) and Davis et al. (2008) because of the broader scope of this research (which included upstream energy usage associated with grain).

The study was based on a series of desktop studies for feed grains and forage. Considering their importance, further research in this area is warranted to ensure correct data are used.

When the full supply chain was considered for context, PE was found to range from 14.3 MJ / kg HSCW for supply chain 1 (short fed) to 32.2 MJ / kg HSCW for supply chain 2 (long fed). These results were generally comparable to previous Australian research and the literature.

10.1.3 Water usage

Blue water usage was 871 L / kg LWT gain for feedlot 1 (short fed) and 151 L / kg LWT gain at feedlot 2 (long fed). Water usage at the feedlot was dominated by water carried through with irrigated commodities (95% of the water usage for feedlot 1). Direct water usage at the feedlot was a smaller contributor. These results are considerably higher than previously estimated because of the inclusion of water associated with irrigated commodities and water captured at the feedlot in effluent holding ponds. Though subject to debate, the inclusion of water from these sources is in line with recent advances in LCA and water footprinting research. This research suggests that regardless of source, all blue water resources should be assigned to the product that consumed them or caused the usage (which includes unavoidable losses and blue water capture for environmental purposes, such as feedlot effluent holding ponds). The rationale is that, for water captured on the feedlot site in effluent containment dams, this water is being restricted from flowing to natural water courses or other users and should therefore be attributed to the production system.

When the full supply chain was considered for context, blue water usage was 460 L / kg HSCW for supply chain 1 (short fed) and 222 L / kg HSCW for supply chain 2 (long fed). These values were within the range suggested by Peters et al. (2009a). When blue and green water were combined (comparable to the virtual water or water footprint of beef) the water usage was 21,606 L / kg HSCW for supply chain 1 (short fed) and 18,612 L / kg HSCW for supply chain 2 (long fed).

Previous research (Peters et al. 2009a) suggested water usage for beef production was as low as 27 L / kg HSCW, which is clearly well below the drinking water requirements of a beef herd. This was because the method applied did not require accounting of water that is not 'pumped', thereby excluding water captured in farm dams for example.

As highlighted by Wiedemann et al. (2010a), water usage is likely to be higher than previously estimated by Peters et al. (2009a) because of the continued use of irrigation for feeding cattle, and from the use of irrigated commodities as supplements or feedlot rations. However, further research is needed to gain more representative data on this.

10.2 Knowledge gaps

10.2.1 Manure greenhouse gas

This study has focussed on GHG emissions from the manure stream at the feedlot and will not extend comments to enteric methane (which has recently been reviewed for MLA in project B.CCH.2022).

For manure emissions, the most important knowledge gaps relate to the nitrogen and carbon cycle at the feedlot pad and manure storage. From mass balance principles, the total amount of nitrogen lost from the feedlot can be reasonably estimated, but the *forms of nitrogen (nitrous oxide, ammonia etc) that are lost are not well understood.*

To date there has been little research to validate the mass flows of nitrogen at the feedlot, which limits the current understanding of potential emission sources. This project has generated a basic theoretical mass balance, but more research is required to substantiate this with the available literature and available feedlot waste stream data. This is required to provide a robust framework for GHG research.

Emission factors for several important manure GHG's are poorly quantified in the literature and are difficult to relate to Australian conditions. In particular, there is insufficient data available for of **nitrous oxide from the feedlot pad and manure storage area** (which are currently grouped by the DCC emission factor).

It is imperative that further research relate nitrous oxide to excreted nitrogen from the livestock for both the estimation of pad and stockpile emissions, and that measurements follow the nitrogen flow (i.e. mass balance) through these two stages of the system. This is required to update the DCC calculation method.

Research gaps are also evident for ammonia emissions and emissions of nitrous oxide from ammonia deposition. These emissions need to be researched concurrently, and must be related to the mass balance of nitrogen at the feedlot for use in the DCC methodology.

There is a need for further research on nitrous oxide emission factors for manure application, which are considered too high for Australian conditions. However, this will be of lesser importance if the ammonia emission factors are altered.

Manure methane emissions are also poorly understood at the feedlot pad, where they may contribute up to 10% of manure emissions. This is particularly relevant for Queensland, where a higher methane emission factor (5%) is applied compared to the southern states (1.5%).

10.2.2 Water and energy usage

The primary knowledge gaps related to water and energy usage are related to upstream processes, particularly commodity production (water and energy) and the prevalence of irrigated pastures in upstream beef production. Rapid assessments of water usage from ABS data suggest that the contribution may be higher than previously thought, and it is clear that even a small proportion of commodities from irrigated systems within the ration can greatly affect feedlot water usage. Likewise, the assessment of energy usage in this study found upstream grain production to be the most significant impact area for feedlot beef production.

This highlights a recognised knowledge gap in feed grain and fodder LCI data. This is relevant for GHG and energy assessment also, and has been identified by several industries interested in LCA research.

10.3 Research recommendations

10.3.1 LCA research

While only preliminary in nature, this study represents a more comprehensive analysis of feedlot GHG, water and energy than has previously been completed in Australia. Considering the variability found by previous data collection studies (Davis & Watts 2006; Davis et al. (2008), it would be highly beneficial to expand the scope of this study to an additional two feedlots (domestic and mid-fed) where data are available (from FLOT.328 / B.FLT.0339) to examine the impact of different feeding time, rations and regional conditions such as climate. This would generate a more robust and representative assessment of the industry. Considering the higher GHG emissions found in this study, it would be valuable to investigate offsets that could be gained by substituting manure for fertiliser in the system.

Upstream grain and fodder production was found to be important for GWP (25-30% of burden), energy (89-90% of burden) and water (up to 95% of blue water burden). Consequently, further research is required for feed grains such as wheat, barley, sorghum and soybean. It is recommended that MLA promote this research with GRDC or investigate independent co-funding options with other RDC’s to speed the supply of these data.

Research is also required to investigate fodder crops relevant to the feedlot industry. There may be options for co-funding of this research with RIRDC.

10.3.2 Manure GHG research

MLA has identified GHG emissions from manure as a priority for research. This project has collated current literature for manure emissions into a theoretical mass balance, which has been integrated into the LCA to assess the relativity of emission sources. The theoretical mass balance was also compared to the current recommended DCC emission estimation approach. This has led to a prioritisation of emission sources and factors (Table 37) which clearly identifies the importance of feed pad and stockpile nitrous oxide emissions.

Considering the DCC (2007a) is the recognised method for estimating manure emissions, research attention should focus on areas that are significant for the current methodology, while also moving towards a longer term goal of establishing improved estimation methods throughout the manure management system.

From this review and assessment, there are important recommendations relating to the research framework. **For emission research to be applicable to the DCC and IPCC methodologies, research must be done with careful reference to the mass flows of nitrogen and volatile solids through the feedlot**.

Further research is required to produce a comprehensive literature mass balance for nitrogen and carbon (volatile solids) in the feedlot system, from feed intake to end use. This mass balance can be validated with data collected for manure and effluent nitrogen to quantify total losses.

For accurate emission estimation, validation of a full mass balance model will be required to relate emission measurements at the **feed pad and stockpile to excreted N from cattle**. To achieve this the following data are required (at a minimum) in addition to nitrous oxide emission measurements:

- Ration nitrogen % and dry matter digestibility.
- Cattle liveweight, average daily gain, feed consumption and feed conversion efficiency.
- Excreted N (estimated, possibly validated with met-crate research).
- Days on feed in feedlot pen and manure deposition rates.
- Feed pad manure deposition variability data (to ensure accurate measurements are collected across the pad).
- Daily climate data from feedlot weather stations.
- Ammonia losses from the feedlot pen (measured data).
- Nitrous oxide emissions from the feedlot pen.
- Mass flow estimates following pen cleaning.
- Nitrous oxide emissions from stockpiles, **following the same nitrogen flow as the feedpad measurements**.
- Ammonia emissions from stockpiles, **following the same nitrogen flow as the feedpad measurements**.

Without this framework, measurement campaigns will be difficult to relate to the DCC estimation model and are at risk of being invalid because of variability in diets, climate and pad conditions. It is noted that a mass balance framework this is the only approach recommended for tier three measurement techniques of manure emissions as listed by the IPCC (Dong et al. 2006).

Emission research is required for feed pad methane emissions, particularly for Queensland feedlots where a higher emission factor is recommended by the DCC.

Considering both nitrous oxide and methane are generated at higher rates in wet, warm conditions, research will need to be carefully linked to climate data to generate an accurate emission profile over a given feeding period.

To address knowledge gaps with manure application nitrous oxide and atmospheric deposition; it may be more appropriate for MLA to commission smaller scale research projects to investigate these emission sources and develop new emission factors, which are expected to be lower than current DCC default values. This research would be supported by the weight of evidence for other emission sources (i.e. fertiliser N) suggesting considerably lower emission rates under Australian dryland cropping conditions, where most manure is used. This is also in line with the IPCC methodology (De Klein et al. 2006) which does **not** recommend a separate emission factor for manure compared to fertiliser.

Considering the lower relative importance of other emission sources such as effluent ponds (based on the theoretical mass balance), these areas are considered a low priority at the present time.

Manure carbon sequestration was found to have a relatively small contribution to offsetting manure emissions in this study (6%), but it is noted that the variability in sequestration rates is high. Considering the lack of literature for carbon levels in soils spread with manure *at appropriate rates*, a pilot study investigating soil carbon levels in manure application areas at two to three feedlots may be beneficial. This would identify if sequestration is likely to be occurring before more detailed research is undertaken, and could be undertaken in conjunction with nitrous oxide research in manure application areas.

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Appendix 1

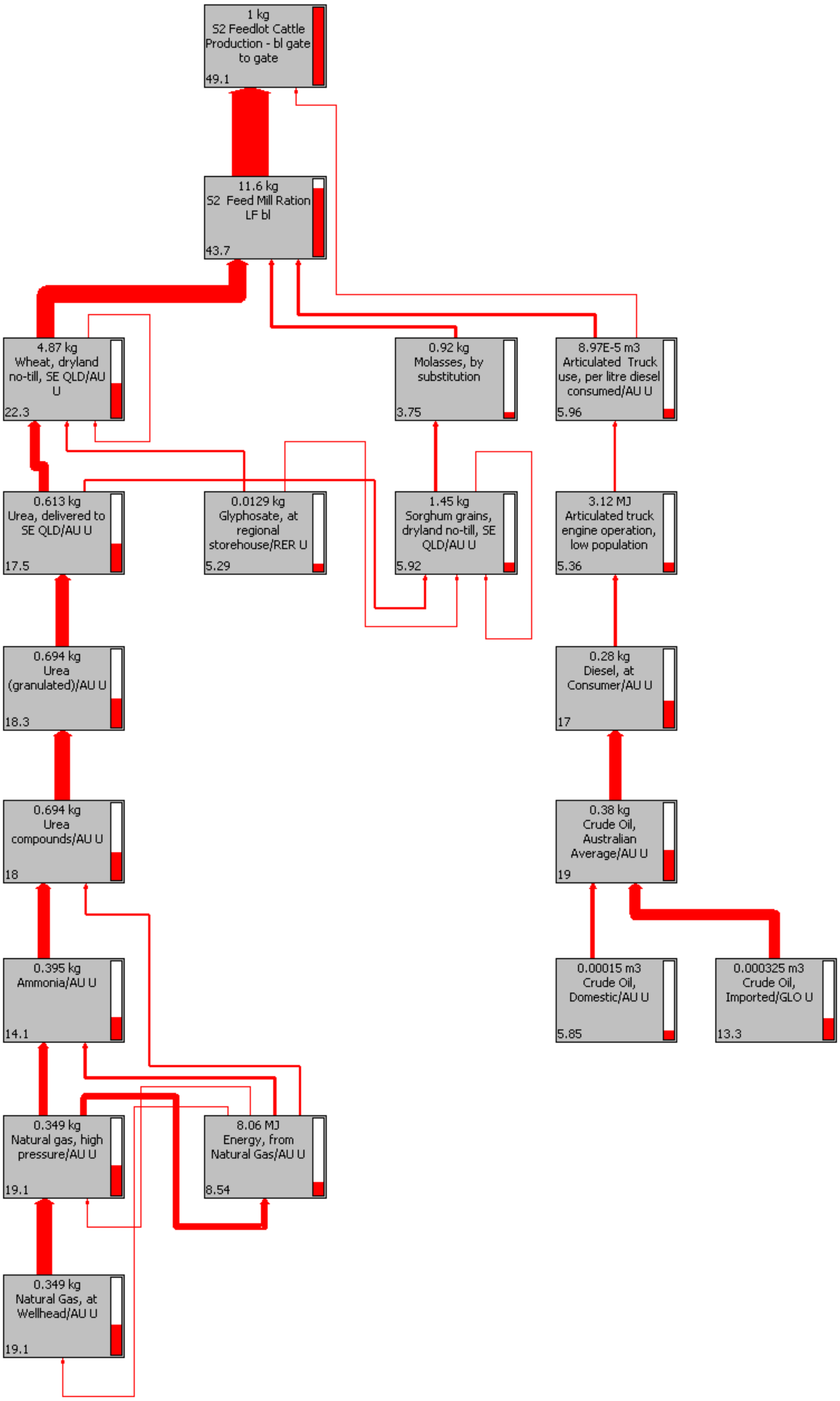

FIGURE 22 – PRIMARY ENERGY USAGE FOR LONG FED BEEF CATTLE PRODUCTION AT THE FEEDLOT (SUPPLY CHAIN 2 – PER KG LWT GAIN)