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Best practice management to increase the profitability of Merino sheep meat production

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Abstract

Greenhouse gas (GHG) emissions and carbon sequestration related to red meat production arise from a multiplicity of sources throughout the supply chain, with each being governed by specific conditions related to animal production, soils, manure and effluent, vegetation and fossil fuel energy usage. All these factors are influenced by variations in climate and management. As yet, many of the specific scientific research areas are still under development or are yet to be investigated under Australian conditions.

With such a broad scope, estimating emissions and defining research targets is a challenging task for the industry. This is further complicated by the range of estimation frameworks that are used by different sectors for their own purposes. Moreover, because GHG emissions are very difficult and costly to measure, most of these frameworks depend on estimation equations and emission factors that are inflexible and not specific to Australian climate or management conditions.

This review covers methodological issues related to GHG estimation in the red meat supply chain and reviews the literature available for GHG emissions from red meat and alternative protein sources. Two additional technical reports cover the specific issues of enteric methane generation and mitigation (report 2), and emissions related to soils and manure (report 3).

To address the broad scope need for GHG estimation throughout the industry, life cycle assessment (LCA) is recommended as a research tool that has the ability to estimate all emission sources throughout the supply chain, providing reasonably robust results at the product (per kilogram of red meat as Hot Standard Carcass Weight - HSCW) or business level if required. The data required for LCA may be used to provide assessments under alternative assessment frameworks such as the NGERS or proposed CPRS if required. LCA can also quantify industry emission 'hotspots' and show the potential of mitigation techniques to reduce overall emissions, allowing both industry members and researchers to understand where attention should be placed and the likely gains that could be made.

As with GHG emissions, water usage in also an important issue for the red meat industries. Calculation of water usage associated with red meat production is surprisingly complex and plagued by inaccurate data in the media and even within the peer reviewed literature. This review concludes that most inaccuracies relate to poor methodology or ambiguous definitions for water use leading to erroneous conclusions from research. There is on-going confusion between traditional water usage from surface or groundwater sources (so called 'blue water' use) and water usage figures that include rainfall to some degree (virtual water - VW). 'Green water' has been proposed as a new descriptor for soil evapotranspiration water derived from rainfall in order to help refine water usage estimates, though at this stage few VW studies differentiate between blue and green water. These form the extremes of the water usage methodologies and subsequent results, explaining why water usage for beef can vary from 27 L to over 200,000 L / kg HSCW.

Water estimation methodology is improving in this field and is at the point where hybrid methods could be applied to Australian case studies in the red meat supply chain to improve the quality of results available to the industry and public. Life cycle assessment is recommended as the overarching framework to achieve this because of the dual focus on both resource usage and environmental impacts. LCA could be used to achieve a new approach that incorporates rigorous water balancing for the calculation of 'blue' water along with estimation of green water requirements. Results can then be analysed to determine the likely impact of water usage on aquatic environments and to identify water hotspots within the supply chain.

Executive Summary

Greenhouse gas emissions and mitigation options are extremely diverse throughout the red meat supply chain. The fundamental processes driving greenhouse emissions and mitigation options range across the traditional fields within agricultural science and engineering. For the industry to respond and adapt to the changing regulations related to greenhouse emissions, a robust framework needs to be established that can account for this diversity. Similarly, water usage in red meat production can be estimated using a surprising variety of methods that produce results from 27 L to 200,000 L / kg beef. Clearly for the industry to address water usage (a resource issue) and water usage (as a *possible* environmental impact) a robust methodology that suits the industry needs to be adopted. This report, the first of three to be presented for the project, covers methodological issues related to greenhouse gas estimation at the industry level, energy usage, a review of vegetation management regulations, and a review of the literature on greenhouse gas emissions and water use for red meat and alternative protein sources.

GHG Accounting

This review ambitiously aims to cover all aspects of greenhouse gas emissions throughout the red meat supply chain, from the farm, feedlot and processor sectors. Reports two and three divide out the two areas where the greatest number of technical research questions need to be addressed. These are: Report 2 – enteric methane research and mitigation, and Report 3 – soil nitrogen and carbon cycling emissions (as related to livestock excretions). The project begins the process of integrating research on greenhouse gas (GHG) emissions and water usage into a single, comprehensive framework that will provide an industry wide context for results. Following a review of greenhouse gas accounting frameworks, life cycle assessment (LCA) is recommended because of the comprehensiveness of this framework and the inherent focus on production efficiency through the use of a functional unit. As the most comprehensive GHG assessment framework available, data collected for an LCA project can be analysed under most other frameworks listed for GHG assessment. LCA can be used both to guide future research and to quantify the performance of Australian produce. This provides a robust basis for promotion and marketing based on environmental credentials. LCA is also able to encompass other environmental impact areas such as energy and water usage which are covered in this report.

To progress research in the field of LCA, standards and guidelines need to be improved for future practice. This may be done internally or as part of cross-industry initiatives such as the work being promoted by the RIRDC. The key issues which should be considered as part of a rigorous methodology for on-farm GHG assessment in LCA research should include:

- The use of representative and comparable supply chains.
- A combination of detailed farm-level data augmented with national/industry wide data to improve the representativeness of the datasets.
- Logical scenarios and sensitivity analysis of key emission hotspots based on the most up-to-date research, particularly in the fields of enteric methane emissions, soils and manure emissions.
- Data collation that is flexible enough to allow analysis under alternative assessment structures e.g. regulated government schemes such as the NGERS and proposed CPRS.

GHG Emissions from Energy Usage

Energy consumption contributes a relatively small component (approximately 5%) of the total supply chain greenhouse gas emissions for red meat production. However, reducing energy consumption will offer many sectors in the supply chain significant cost savings along with the minor contribution towards reducing emissions.

Some sectors of the industry already come under government regulations for resource efficiency assessments. Hence, the industry needs to identify and improve the uptake of cost-effective energy efficiency opportunities. As the first step, this should include an enterprise level (farm or facility) energy audit to identify energy and cost saving opportunities and highlight potential improvements in productivity and quality. The outcomes of work in the feedlot sector (Davis et al. 2008) have included benchmarking energy usage at an enterprise and sector level and the development of a framework for energy monitoring and evaluation. The mechanisms underpinning this framework and the methodology may be useful for the grazing and processing sector.

Technologies for converting organic by-products to energy are advancing rapidly and offer one of the greatest opportunities to decrease reliance on fossil fuels and reduce greenhouse gas emissions at the same time. There are now many facilities operating or under development and planning not only overseas but also in Australia. This is seen as the best strategic abatement option for energy consumption available to the intensive sectors of industry. To date, development has been inhibited because of the low cost of energy (low incentive) and the lack of mechanisms to control demand, but this is now shifting with a series of new government regulations that favour renewable energy and may lead to increased costs for fossil fuel energy.

It is recommended that a program be implemented to clarify the current NGERS policies and methodologies from the perspective of the red meat industry. This program should:

- Increase awareness within the red meat industry of its obligations under current legislation and proposed legislation,
- Assist the respective industry sectors to meet their obligations under current legislation in measuring and reporting emissions,
- Develop tools to facilitate the abovementioned elements.

It is also recommended that a program be implemented to identify strategies to reduce demand and consumption of energy and to ensure that energy consumed is produced from lower greenhouse intensive sources. In this regard expanding the use of renewable energy is critical.

The potential for renewable energy sources is clear within the feedlot and meat processing sectors (energy generation from waste streams) and this needs to be promoted at an industry and government level.

Processing sector

Meat processing plants generate large amounts of greenhouse gases from energy usage and waste treatment. The main GHG mitigation option apart from traditional energy use saving approaches will focus on methane capture and utilisation. The best options to achieve this are likely to be either; i) covering ponds with impermeable high density polyethylene (HDPE) covers and flaring methane (low cost), or ii) building purpose built digesters to generate methane that can be used for heat or electricity generation (high cost). Other mitigation options for the sector may come from reducing the organic load in the waste stream or researching emissions from highly loaded, crusted ponds which may show lower emission levels. At this stage research in the area is lacking in the publically available literature. Future research may be an opportunity for collaboration between the processing sector and the feedlot sector because of the similarities in treatment options.

Reducing the quantity or organic matter in the meat processing waste stream will also reduce the amount of methane generated due to lowering the amount of chemical oxygen demand (COD) entering the (anaerobic) waste treatment system. This can be achieved through the further removal of solid material before wastewater. Ways of reducing organic load include; screening of individual wastewater streams to recover lost product, segregation of hot water streams to improve fat recovery and removal of stick-water solids using evaporation. These options are discussed in further detail in the Eco-efficiency manual for meat processing (MLA 2002).

The emissions from meat processing waste treatment ponds have not been investigated extensively in the literature, and further research to validate emission factors under a range of management scenarios may be warranted. Likewise, emissions from nitrous oxide in meat processing waste streams have not been quantified and may not have been included in industry wide research such as the life cycle assessment project. Hence, further research is required to quantify these emissions and place them in the context of the wider industry.

It is recommend that a program be implemented to clarify the current government GHG methodologies (the DCC manual) and policies (i.e. the NGERS) from the perspective of the processing sector. This program should include:

- Fundamental research to characterise GHG emissions from meat processing waste streams under a variety of current and potential waste treatment strategies,
- Development of tier 3 methods for alternative waste management strategies that may be used by the Australian meat processing sector where significant differences in emission levels are identified,
- Pathways to increase awareness within the processing sector of its obligations under current legislation and proposed legislation.
- Assisting the processing sector to meet their obligations under current legislation in measuring emissions and reporting emissions.
- Development of tools to facilitate the abovementioned elements.

It is also recommended that a program be implemented to identify strategies to reduce demand and consumption of energy and to ensure that the energy that is consumed is produced from renewable energy as much as possible.

Vegetation Management and Regulations

Land use and vegetation management with respect to carbon emissions and sequestration influences the red meat industries from two directions, i) through the effect of legislation on grazing lands which may lead to lower grazing productivity, reduced land value and an increased management and regulatory burden for farmers because of increasingly stringent vegetation laws, and ii) through the opportunities that land managers have for offsetting livestock emissions through sequestration in carbon.

The red meat industry needs to address both of these very different challenges through engagement with the public and the government. Considering the status of the industry with the vegetation laws that have already been introduced, it is important that the industry promote the role livestock producers are already playing to reduce national emissions and lobby for better recognition and financial offsets for landowners based on this role.

Because of the key role that land use emissions and sequestration opportunities are seen to have in enabling Australia to meet its Kyoto objectives and because of government policies to preserve biodiversity and ecosystem services, this area has been subjected to more and more stringent legislative controls across the country. As a result, vegetation (and the carbon sequestered) has largely become a privately managed public asset. This has reduced the opportunity for land managers to use these assets for the benefit of their own business. New vegetation laws introduced in Queensland controlling regrowth are an instance where legislation is likely to decrease stocking capacity and could transfer carbon sequestration rights from the landowner to the Queensland Government for timber re-growth on previously cleared land.

Despite this, the red meat industries may still have the capacity to receive credits for carbon sequestration to offset emissions where land has been cleared prior to 1990 in most cases. Hence it is in the long-term interests of the red meat industry to investigate and implement effective practices to enhance the amount of carbon sequestered through vegetation management. To date it is not clear how much carbon could potentially be sequestered in Australia's rangeland areas, though estimates are as high as half of Australia's annual emissions. However, the level of carbon sequestration that could be achieved on livestock properties *without significantly decreasing productivity* has not been quantified to date and may be quite low in some regions.

To provide an indication of land areas required to offset annual livestock emissions with agro-forestry, a number of case studies were developed for grazing regions around Australia. The case studies show that 9% to 104% of total land area would need to be planted to trees in order to offset livestock emissions for the two beef case study properties, while between 6% and 66% of total land area would need to be planted for the two case study lamb producing properties. These scenarios were based on a 70 year period. Sequestration through agro-forestry was generally more realistic in high rainfall regions (northern NSW case study) than in lower rainfall zones (i.e. southern NSW case study). Detailed agro-forestry assessments were not available in the literature to verify these case studies however, and further research would be required to confirm these trends.

The likelihood of producers needing to mitigate livestock emissions is still unknown, as is the potential for the government to provide free carbon emission permits. If livestock managers were given substantial proportions of free permits (up to 90%), sequestration may allow the remaining emissions to be effectively offset. On some farms this would not only be achievable, but may have benefits to the production system through provision of shelter for livestock and salinity control in some regions. Tree plantations may provide ecosystem services or possibly diversified income through agro-forestry.

Land managers will choose among mitigation options depending on the degree of vegetation on their land already, possible tree growth rates (sequestration rates / ha), carbon prices, and the development of new markets for timber products (for example, for biofuels).

It is recommended that a program be implemented to address knowledge gaps identified in this area, which include:

- Assessment of the real and potential impacts of vegetation management laws on the viability of grazing enterprises (reduced carrying capacity, reduced land value, loss of carbon sequestration rights)
- Quantification of the value of reduced emissions and sequestration from the grazing sector to the Australian community both on an industry basis and on a 'per kilogram of product' basis to overcome the division of land use and livestock emission estimation
- Further quantification of potential and actual sequestration rates for agro-forestry (tonnes CO_2-eq / ha / year) for a range of livestock production regions in Australia and determination of land areas and economic implications from offsetting livestock emissions through on-farm agro-forestry.

- Research and policy development to ensure fair treatment of on-farm sequestration, including research to develop cost-effective sequestration measurement techniques.

Alternative Protein Sources

A review of LCA research for red meat and alternative proteins suggests that Australia has the potential to produce environmentally efficient red meat with superior performance to many countries in the world. While not extensively supported by Australian research (there is only one LCA study to date for Australian red meat), there are underlying factors that promote this, such as the low intensity of energy use, and the use of non-arable land and minimal grain feeding.

Many studies have presented meat, and particularly red meat as the ‘worst’ protein product from an environmental perspective, particularly with respect to greenhouse gases. This has led to calls for reduced consumption of meat products, particularly in Westernised countries.

The LCA literature indicates that red meat production generally produces more GHG per kg of product or per meal than white meat or plant protein alternatives, and is likely to have similar emissions to high protein dairy products such as cheese. Many of these studies however do not allocate GHG emissions to animal by-products. It is noted that pork and chicken meat production result in less valuable by-products than sheep and cattle production (e.g. wool and leather). This makes the handling of co-products very important to the conclusions of comparison studies (Garnett 2009).

While there are some underlying issues related to red meat production that disadvantage these species in comparison to other animal species (i.e. the low breeding rate and ruminant digestion system), the comparative advantage of the ruminant digestive system (the ability to produce protein from low quality forages) is rarely taken into account. This ability means that, from a land use perspective, very few animal or plant products can be produced on the types of land that can be used for red meat production. Studies in Europe have not been well positioned to take this into account, as a high proportion of red meat production is reliant on grain and therefore arable land. Only one study (Williams et al. 2006) investigated two types of land to improve the assessment of grazing animals in their study, though the results were still skewed by the use of grains fed to livestock.

Another factor in comparisons is the reliance of different industries on energy. The international literature often shows red meat as energy intensive to produce (i.e. Ogino et al. 2004; Weidema et al. 2008b). While some forms of red meat production (particularly those practised overseas) are highly energy intensive, this does not need to be the case, as demonstrated by Peters et al. (2009) for extensive, Australian red meat production.

The perspective on red meat will be driven by what is considered the most limiting resource or environmental factor. If greenhouse gas emissions are the limiting factor, red meat will need to make large gains in performance and is unlikely to be comparable to other meat products or plant alternatives. This may cause beef and sheep meat production to decrease where alternatives become more profitable (i.e. if carbon emissions are taxed), making the production of feedlot beef, for example, less competitive. This may lead to a shift in red meat production to non-arable land that is not suited to other forms of agriculture. With the current knowledge of enteric emissions, this would lead to higher emissions per kilogram of product compared to intensive production (such as lot feeding). If profitability is too low from extensive rangeland production, this land may be removed from traditional agricultural production, leading to an overall reduction in food production from Australia. This highlights the tension between the goals of food production and reduction of greenhouse emissions. A similar tension exists in the biofuels production debate between emissions reduction and land use for food production (Wiedemann et al. 2008).

Because of the unique production practices carried out in Australia and our role in international trade, there is may be an opportunity for the red meat industries to substantiate their claim as being low impact, resource efficient products. However, this assessment would be better made by investigating a wider number of environmental issues than simply GHG emissions. This would require validation of Australian LCA results using a wider assessment of the industry, as has been done by Williams et al. (2006) for the UK. This project has been highlighted in report 2 of this project as a possible model for establishing a co-ordinated Australian research program for red meat. A similar Australian project could carry dual roles for research and promotion of Australian product.

It is recommended that a program be implemented to address knowledge gaps identified in this area. This may be done by commissioning an LCA project (or projects) with the following objectives:

- Provide comprehensive Australian LCA research across all industry sectors using representative farms and covering the northern beef sector.
- LCA research on the greenhouse gas emissions energy usage required for Australian grain production in the eastern states where grain is used for feedlot beef production.
- Comparison of competing products that compare ‘like with like’ – i.e. Australian and US grain-fed beef for the Japanese market.
- Compare land use for red meat production and alternative protein sources identifying requirements for non-arable, arable, and irrigated land.

It is important that further LCA research implement a scientifically rigorous methodology for GHG estimation at the farm, feedlot and processor level, incorporating up-to-date research on key emission sources in addition to the NGGI (Department of Climate Change) methodology. Key emission sources include enteric methane, soil and manure emissions and land use change effects. The research should investigate logical mitigation scenarios (based on the results of recent scientific research) and determine the sensitivity of the overall emissions based on a review of scientific literature. A project of this nature would allow dissimilar emission sources (i.e. enteric methane and soil nitrous oxide) to be placed in the overall context of red meat production. Likewise it could allow research and mitigation options to be compared and categorised into most effective to least effective.

Water Usage Methodology Review

Water use methodologies were reviewed to provide a clear understanding of the strengths and limitations of the various methodologies available. The methodologies can be broadly grouped into three categories – water engineering, virtual water / water footprints and LCA. Whilst they have been developed for different purposes and may relate at some levels, they rarely produce comparable results and are often used to draw conclusions that cannot be substantiated from the methodology used. In particular, few methods have been developed to show the *environmental impact* of water use. Rather, the implication is made that ‘the more you use the worse you are’. This reflects the traditional focus on ‘resource use’ rather than ‘impact of use’ and is clearly inadequate for the latter.

The traditional approach to water use assessment adopted by private enterprises and governments is to define the quantity of water used in a particular locality (i.e. a farm, catchment, state) using a water balance. Farm water balance estimates are typically made using models of hydrology and crop production, and these can be used on a broader scale.

The strength of this approach – when used for water accounting – 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. This approach has been used successfully in the feedlot and processing sectors to estimate water usage.

Based on the ABS definitions, water usage estimates for Australia's beef industry (from point source property data or a broad scale economic assessment) range from 27 to 540 L/kg HSCW. However, when first order estimates of the contribution of irrigation water to feed inputs for beef production (pastures and grains) were determined, the water usage estimate was 474 L / kg HSCW as a national average (i.e. water used for irrigation was divided by total Australian beef production). This suggests water usage for beef production may be on the higher end of the range estimated by Peters et al. (2009a).

Many assessments of water usage for red meat production have been made using the virtual water and water footprint methodologies. These estimates vary greatly from 15,000 to 200,000 L/kg of beef and 6,000 to 51,000 L/kg of sheep meat. In most studies, the system boundary is unclear and water from both stored sources (i.e. dams, rivers, and groundwater – designated 'blue water') and soil stored rainfall (designated 'green water') are included with no distinction between them. While specific water footprint studies differentiating between blue and green water for Australian red meat are not available, the contribution from blue water is expected to be very low (≤ 3%).

From a virtual water (VW) or water footprint perspective, meat is a more 'water intensive' product than most plant products and these results have been used as an argument to promote reduced meat consumption. However, without knowing anything about the form of water used (blue or green), the land used in the production of the product (arable or non-arable) or other contributing factors, it is impossible to state that reducing consumption will result in genuine water savings or, by implication, benefits to the environment.

As a trade tool for alleviating water stress by trading 'embedded' water with products, the virtual water concept has merit. However, as a proxy for the environmental impact that water usage has on aquatic environments (i.e. rivers), the concept is misleading when no differentiation of the source of water (blue or green) has been clearly elaborated and correctly interpreted in the results and discussion.

Therefore, the VW and water footprint concepts in their current form are not able to provide adequate detail to be of value in environmental assessments of water usage in red meat production in Australia. However, in as much as the industry is a supplier of food to the world, these concepts may be useful and of interest into the future. There is a need for researchers to clearly state the limitations of their research to avoid misrepresentation of data in the media.

Having reviewed the water balance and virtual water methodologies, LCA is considered the best over arching framework with which to study water use for the industry. This is because LCA has the ability to integrate useful methods for water use estimation from both of the other concepts while presenting data on a functional unit basis. LCA also covers other issues of importance such as GHG emissions discussed previously.

The following knowledge gaps have been identified in this literature review:

- Detailed water use inventories for red meat that specify water by source and by type (blue and green water),
- Detailed water use inventories for major commodity inputs to red meat production such as grains and fodder (blue and green water),
- In depth review of broad scale Australian water use data (ABS and catchment scale water balances) to improve estimates of water use in the red meat industry.
- Research comparing alternative protein sources with a methodology (as identified in the review) that allows reporting of blue and green water, and the impacts of water use on the environment.

A methodology has been proposed to improve the assessment of water usage in the red meat industry. The exact approach taken will depend on the future goals of the industry, however it now appears that the required methods are available in the field of LCA and through hybrid approaches using the methods of water engineering (at the farm and catchment scale) and virtual water (for determination of embedded water from inputs into the agricultural system).

It is recommended that the industry conduct case studies to test the application of a more detailed LCA approach with water use impact assessments within the beef and lamb industries. A first step would be to utilise the data already collected in various Meat and Livestock Australia (MLA) projects and re-analyse these using Australian Bureau of Statistics (ABS) regional water use and production data for irrigated pastures and crops. Results could then be presented for blue and green water usage. The identified knowledge gaps could be addressed through a LCA project or series of projects with similar objectives to those identified for GHG.

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List of Abbreviations

ABARE – Australian Bureau of Agricultural and Resource Economics

ABS – Australian Bureau of Statistics

AGO – Australian Greenhouse Office

CFCs – Chlorofluorocarbons

CH_4 – Methane

CO_2 – Carbon Dioxide

COAG – Council of Australian Governments

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

GABSI – The Great Artesian Basin Sustainability Initiative

GHG – Greenhouse Gas

GHGP – Greenhouse Gas Protocol Initiative

GM – Genetically Modified

GWP – Global Warming Potential

HDPE – High density polyethylene

HFCs – Hydrofluorocarbons

HSCW – Hot Standard Carcass Weight

IO-LCA – Input-Output Life Cycle Assessment

IPCC – Intergovernmental Panel on Climate Change

LAI – Leaf Area Index

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

NEGRS – The National Greenhouse and Energy Reporting System

NGA – National Greenhouse Accounts

NGGI – National Greenhouse Gas Inventory

NPI – National Pollutant Inventory

NWC – National Water Commission

NWI – National Water Initiative

NVIRP – National Victoria Irrigation Renewable Project

O_3 – Ozone

OA – Organic Agriculture

PFCs – Perfluorocarbons

PID – Private Irrigation District

REC's – Renewable Energy Certificates

SFs – Sulphur hexafluoride

UNFCCC – United Nations Framework Convention on Climate Change

VRD Variable Fixed Drives

VW – Virtual Water

WBCSD – World Business Council for Sustainable Development

WMPP – Wimmera Mallee Pipeline Project

WRI – World Resources Institute

WUE – Water Use Efficiency

1 Project Objectives and Reporting

1.1 Project Objectives

The objectives of this project are to:

- *Conduct a critical review of the literature on the greenhouse gas emissions and water use associated with beef and sheep meat production, including for the on-farm, feedlot, and processing sectors.*

- *Compare and contrast these estimates with those published for beef and sheep meat production overseas, and analyse the reasons underlying any significant differences between these estimates, In particular for water use estimates, discuss the merits and limitations of different methods of calculation in assessing environmental impacts.*

- *Compare and contrast these estimates for Australian red meat production with those published for the major alternative dietary protein sources produced in Australian agriculture, and analyse and describe the opportunities and constraints (economic, agronomic, nutritional, social and environmental) for moving production to those protein sources with lowest environmental impact.*

- *Provide recommendations on:*

  1. *data requirements and understanding needed to improve Australia's greenhouse gas accounts and water use estimation for livestock industries;*
  2. *ways in which the beef and sheep meat industries can significantly reduce greenhouse gas emissions and water use based on current knowledge.*

- *Based on the literature review and analysis, prepare a technical note suitable for publication on the Meat and Livestock Australia (MLA) website, and submit a journal paper for peer review in the third quarter of 2009.*

These objectives will be addressed in the methodology. The outcomes of this work will be the production of a rigorous, holistic and scientifically defensible report that will present:

- A position statement on greenhouse gas production and mitigation, and water use for the livestock industries, including analysis of recent improvements and contributions made by these industries throughout the supply chain,
- Critical analysis of the Australian and International methodologies for estimating greenhouse gas emissions and water use in the livestock industries,
- Critical comparison of greenhouse gas (GHG) emissions and water use for the production of Australian red meat compared with other protein sources (including but not limited to pork, chicken meat and plant proteins),
- An assessment of the comparability of these alternatives with respect to their nutritional properties and the feasibility of producing equivalent volumes of product in Australia or overseas,
- Case study comparisons that compare Australian production with international competitors based on available literature and unpublished data from researchers in this field known to the project team,
- An assessment of the current emissions reporting structures and legislation and the suitability of this structure for the livestock industries (particularly focused on the opportunities for claiming carbon credits to offset livestock emissions),

- Detailed recommendations to improve water use in the Australian livestock industries, covering extensive beef and sheep, the feedlot and processing sectors based on the literature review and current MLA funded projects COMP.094 and B.FLT.0339,
- Recommended research and extension directions that may lead to significant reductions or mitigation options for GHG emissions from Australian livestock production,
- An outline of the data required at the farm level for producers to quantify GHG emissions and water use, allowing improved industry performance to be tracked over time,
- Identification of major knowledge gaps to be addressed through strategic industry and government investment of research and extension funding.

The information will be presented in a peer reviewed publication, a technical note for publication by the industry and a detailed technical report.

1.2 Project Reporting Structure

The report covers a considerable amount of information, particularly in regards to greenhouse gas emissions and mitigation strategies. For this reason, and because of the multiple researchers in the project team, the project will present results in the form of three reports:

1.2.1 Report 1 – Water and Greenhouse Gas Frameworks Review

Report 1 (this report) will provide 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.

The report has been compiled primarily by FSA Consulting staff, edited by Stephen Wiedemann.

An exception is chapter 6 (GHG policy) which is authored by Professors Paul Martin and David Cottle of UNE.

1.2.2 Report 2 – Enteric Methane Review

Because of the significance of this single emission source to the red meat industries, a separate technical report has been completed by Dr David Cottle and Professor John Nolan, covering nutritional and genetic approaches mitigation of emissions from livestock, modelling of livestock emissions and a review of the Department of Climate Change methodologies available for the red meat industries.

1.2.3 Report 3 – Nitrous Oxide and Carbon Cycling in Soils and Waste Review

This report has been completed 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 (intensive and extensive red meat production).

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 processes. 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.

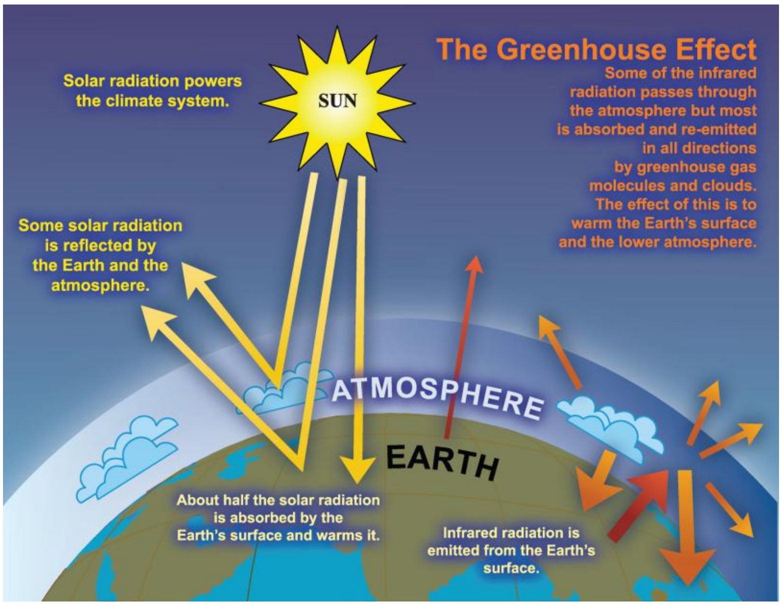

FIGURE 1: AN IDEALISED MODEL OF THE NATURAL GREENHOUSE EFFECT (LE TREUT ET AL. 2007)

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.

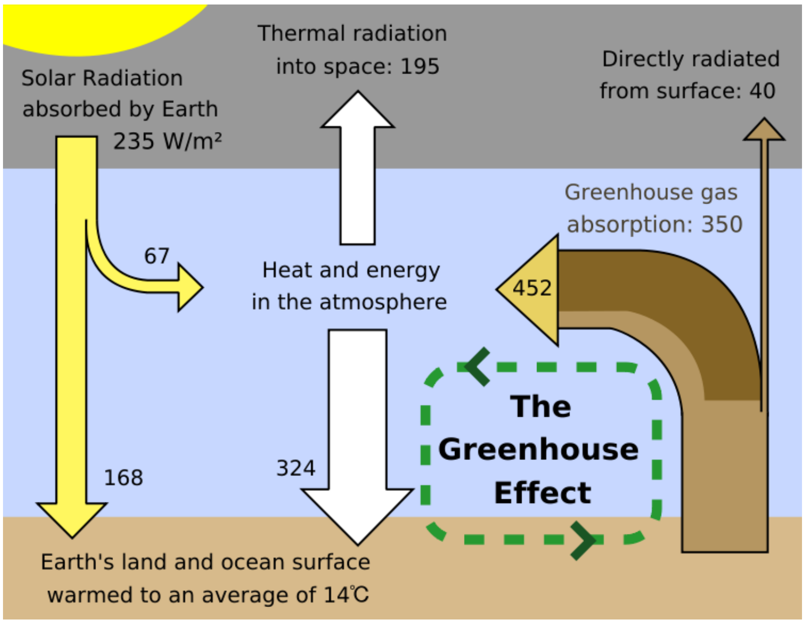

FIGURE 2: THE RADIATION (ENERGY) BALANCE OF THE EARTH (ROHDE 2008).

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 is 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 (IPCC 2007a).

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). Global warming potential depends both on the intrinsic capability of a molecule to absorb heat, and the lifetime of the gas in the atmosphere. The global warming potential values take into account the lifetime, existing concentration and warming potential of gases. Thus, global warming potential 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 global warming potential 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.

Global warming potential 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 1. 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 1: 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: 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 1 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 Gases

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 2 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 2: 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 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 Intergovernmental Panel on Climate Change (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 Department of Climate Change (DCC) (formerly the Australian Greenhouse Office, AGO) provides methodologies for the calculation of GHG emissions for each sector (http://climatechange.gov.au/inventory/methodology/index.html) (Department of Climate Change 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 Co2-e or 16.3% of Australia's GHG emissions, making it the second largest emitting sector behind stationary energy (Department of Climate Change 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 CO2-e (58.9 per cent) and 20.2 Mt CO2-e (85.9 per cent) respectively of the net national emissions for these two gases.

Greenhouse gas emissions from *Agriculture* increased by 1.5 per cent (1.3 Mt) between 1990 and 2007, and decreased by 3.0 per cent (2.7 Mt) from 2006 to 2007. Preliminary estimates for 2008 indicate that Agriculture emissions have increased by 3.0 per cent (2.6 Mt) since 2007 due to increased emissions from savanna burning (Department of Climate Change 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 (DEPARTMENT OF CLIMATE CHANGE 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 (DEPARTMENT OF CLIMATE CHANGE 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 Department of Climate Change uses a GWP for methane of 21 and not 25 for their greenhouse gas inventory calculations (Australian Greenhouse Office 2006a; Department of Climate Change 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 Department of Climate Change uses a GWP of 310 and not 298 for their greenhouse gas inventory calculations (Australian Greenhouse Offic, 2007a; Department of Climate Change 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 Department of Climate Change has developed frameworks such as the National Carbon Accounting System (NCAS) for estimating and reporting greenhouse gas 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 greenhouse gas 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 greenhouse gas (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.

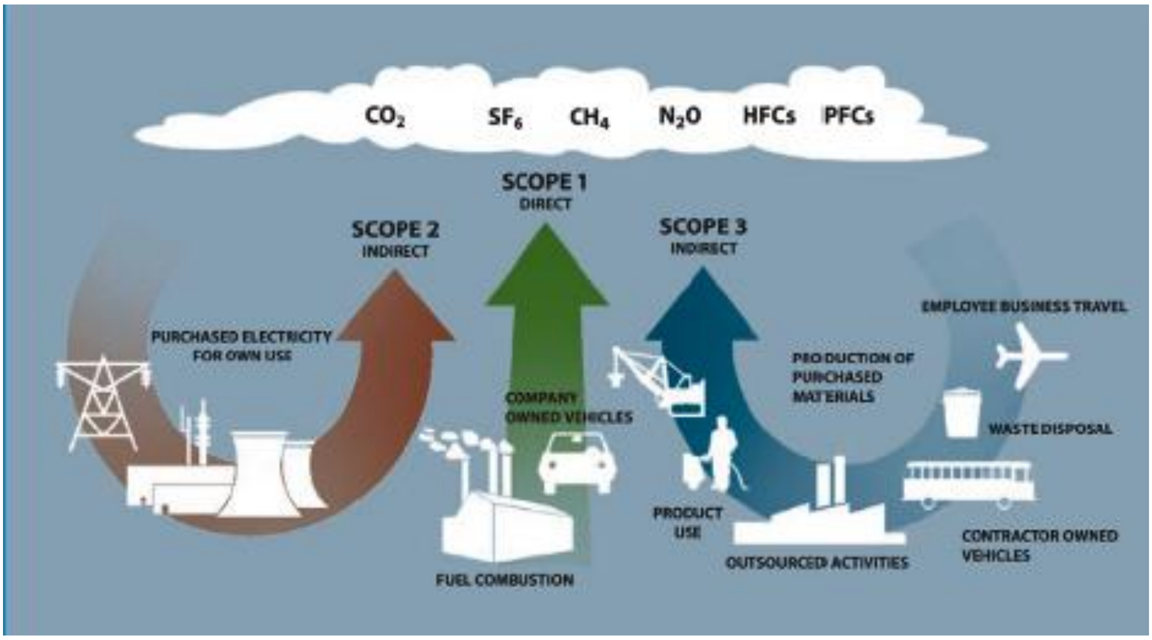

FIGURE 5: EXAMPLES OF SCOPE 1, SCOPE 2 AND SCOPE 3 EMISSIONS (WRI 2004)

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 greenhouse gas 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 NEGR act) establishes a national systematic framework for reporting greenhouse gas emissions and makes registration and reporting mandatory for corporations whose energy production, energy use or greenhouse gas 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' greenhouse gas emissions data will be essential to maintain the environmental and financial integrity of the Carbon Pollution Reduction Scheme (Department of Climate Change 2009a).

The NEGRS 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 greenhouse gas emissions and energy data.

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

Direct and, in some cases, indirect greenhouse gas emission estimates are required to be reported under the NEGR Act. The NGER act classify 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 NEGR 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 (Department of Climate Change 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 (Department of Climate Change 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 “carbon footprint” 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 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 greenhouse gas 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 greenhouse gases 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.

Life cycle assessment (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:

- problem shifting from one stage of the life cycle to another:
- problem shifting from one sort of problem to another: and
- problem shifting 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.

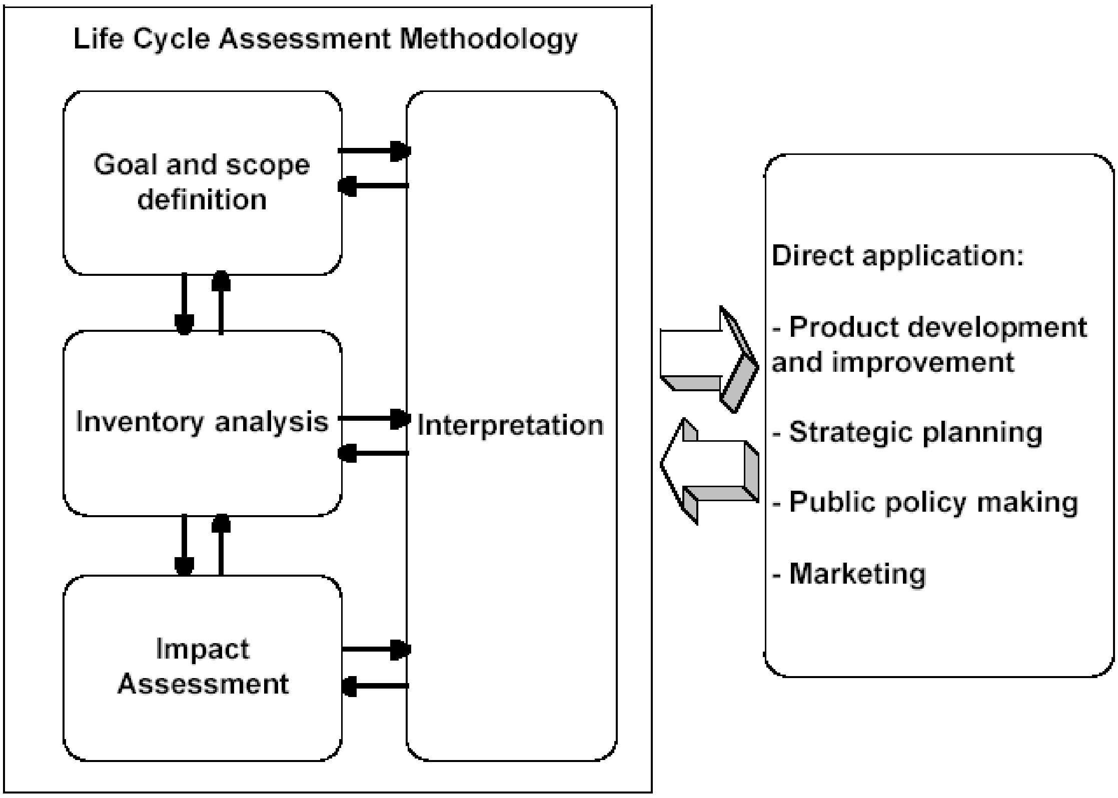

FIGURE 6: GENERAL FRAMEWORK FOR LCA AND ITS APPLICATION (STANDARDS AUSTRALIA 1998)

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 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 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.

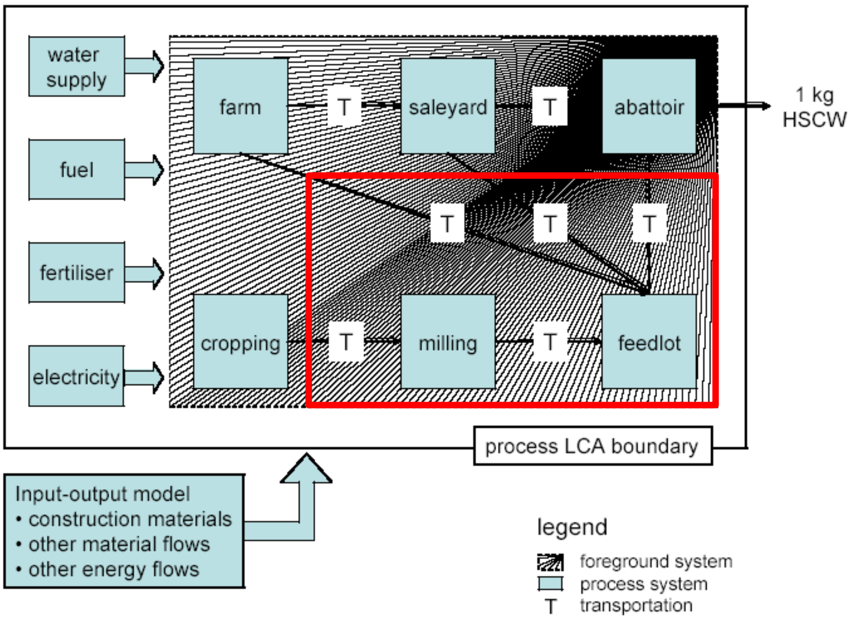

FIGURE 7: GENERALISED RED MEAT SUPPLY CHAIN MODEL WITH FEEDLOT SUB-SYSTEM (PETERS ET AL. 2009A)

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 like any other model, it holds that “garbage in = garbage out” .In other words data quality has a major impact on results and proper evaluation of data quality is therefore an important step in any LCA. The data quality should comply with ISO14040 standards and therefore should include a description of data quality to allow reliability of the results and interpretation of the outcomes to be understood. Harris & Narayanaswamy (2009) and Renouf (2006) outline data types, sources, quality and validation techniques for rural industries.

When considering agricultural products, the majority of impacts (i.e. GHG, water usage, and eutrophication) 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 sciences that drive dominant emissions and resource usage on farms. Moreover, it requires a very good understanding of the agricultural system under study to ensure correct boundaries are drawn and data are collected. 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.

It is commonly the case that at the farm level, a separate model is used to determine major emissions and farm processes. In the Australian work, the Beefbal model was utilised for the feedlot sector of the supply chain, supplemented by real data collected from Australian feedlots (Davis et al. 2008). However, specific emissions such as enteric methane and nitrous oxide, which dominate the GHG profile of red meat, where calculated using the only available literature on these emissions (supplied in the Department of Climate Change 2007 methodology).

It has been proposed by Harris & Narayanaswamy (2009) that, wherever possible, real time data should be collected in conjunction with an LCA. This may not be practical in some cases, as the required research is extensive, but it must be clearly noted that the results of the whole LCA will only be as good as the accuracy of emissions calculation for 2-3 major factors (enteric methane, nitrous oxide and possibly manure methane). Secondly, because an LCA is conducted for a functional unit (i.e. the total kilograms of red meat for a property over a set period of time), the accuracy of the LCA will be greatly affected by the quality of production data that are collected, and the representativeness of these data for the industry.

Input-output LCA (IO-LCA) is a mathematical modelling technique based on a model of the national economy that can be used to ‘fill in the gaps’ of an LCA when detailed process-based LCI data are unavailable (Rebitzer et al. 2002). In process based LCA as much physical data as possible is included (as presented in previous sections). However, it is never possible to include the entire system. For example, how much water is consumed during the manufacturing of vehicle repair materials for the feedlot? Collecting such detailed data would be impractical and expensive. IO-LCA has been proposed as an alternative to conventional LCA, because it overcomes these limitations. IO-LCA involves constructing a mathematical model of the national economy and the environmental impacts of industries. The model can be used to estimate the environmental impacts of any producer based on that producer’s expenditure patterns. However, IOA would not be as accurate as LCA in describing on-farm impacts because of the major role the on-farm processes play in determining environmental impacts (i.e. enteric methane).

A frontier approach to LCA is to combine these two approaches, by using the precision of LCA to get a detailed picture of the main industry being examined, and using IO-LCA to ‘fill in the gaps’ regarding some of the supporting industries. Peters et al. (2009a) developed a sophisticated hybrid model to improve the accuracy of their red meat production LCA by incorporating IO-LCA results into it, though interestingly this did not have a very substantial impact on the final result because of the dominance of the farm level processes. However, the IO-LCA approach does eliminate the two major issues of boundary definition and circularity effects of process-based models.

Handling of co-products

A very sensitive step in the calculation process is the handling of co-products. Most agricultural systems yield more than one product. Therefore, 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 ISO14044 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 done with great care so that the data are representative. The quality aspect is therefore also crucial.

The result of LCA study involving a multi-input/output system is affected significantly by the choice of the allocation method. Peters et al. (2009a) in their LCA made comparisons on the basis of unallocated burdens (no allocation to useful carcase co-products) except with respect to the grain and wool products to allow comparison with the results of different studies. Peters et al. (2009a) used an economic allocation to consider wool by-products for sheep farms.

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, Photooxidant 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 3).

TABLE 3: 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.

3.6 Conclusions, Knowledge Gaps and Recommendations

3.6.1 Conclusions

Life Cycle Assessment is the most comprehensive method for covering emissions from red meat production across the whole supply chain. LCA has an integrated, robust framework, though the application of methodology is diverse in its practice. Methodology for LCA is more clearly defined than for carbon footprinting at this stage. Because of the life-cycle approach, LCA it is able to expand on the business accounting methods, integrating data collected under this framework and providing a 'product' or 'output' result that highlights efficiency of production. For this reason, carefully conducted LCA research can easily provide output aligned with the 'scope 1, 2, 3' definitions for business reporting or industry modelling, whereas the business accounting framework is of limited use at a supply chain or industry level if data are not collected with a broader use in mind.

LCA research has been undertaken in the red meat industry in three major projects (MLA projects COMP.094, FLOT.328 and B.FLT.0339), with the following general aims:

- Assessment of the environmental impacts the industry is having on Australia's environment (including estimation of GHG emissions)
- Provision of defensible water and energy usage data for red meat production that can be used to inform the public, the industry and the government
- Provision of an assessment framework that will be compatible with any government regulations in the future (of particular relevance to the feedlot sector)
- Identification of impact 'hot-spots' within the production system and solutions to improve performance.

There are benefits to the red meat industry from a holistic LCA approach that can be used to overcome the definition problems with the NGGI and "carbon footprint". For example, within LCA there is a method for integrating all the emission sources and sinks (including land use) and providing the results on a 'product unit' basis. This would allow a more holistic assessment of carbon on grazing land that has been carried out to date.

For these reasons LCA is the recommended framework for GHG assessment in the red meat industry. LCA has the capacity to inform the industry of both GHG and other environmental impacts for guiding practice change and research. It provides a context under which all research can be integrated and compared for effectiveness both now and into the future. LCA can also be used to assess industry practice change and policy impacts on environmental impacts and productivity.

3.6.2 Knowledge Gaps and Recommendations

LCA is the most comprehensive methodology for GHG assessment and it is recommended that any further industry level or supply chain research be undertaken within the context of an LCA. Whilst standards have been developed for LCA, it is recognised that there are a number of interpretations and use of these standards in practice. Therefore standards and guidelines need to be improved for future practice. This may be done internally or as part of cross-industry initiatives such as the work being promoted by the RIRDC.

The key issues which should be considered as part of a rigorous methodology for on-farm GHG assessment in research should include:

- The use of representative and comparable supply chains.
- A combination of detailed farm-level data augmented with national/industry wide data to improve the representativeness of each data set.
- Logical scenarios and sensitivity analysis of key emission hotspots based on the most up-to-date research, particularly in the fields of enteric methane emissions, soils and manure emissions.
- Data collation that is flexible enough to allow analysis under alternative assessment structures e.g. regulated government schemes such as the NGERS and proposed CPRS.

4 GHG Emissions from Energy Use in Red Meat Production

4.1 Relationship between Energy Usage and GHG Emissions

Red meat production requires consumption or use of energy in the various stages of the supply chain. A large percentage of the energy used in red meat production is used upstream of the farm or processing sectors in the manufacture of inputs such as fertilisers, pesticides, farm machinery, animal feed, and veterinary drugs.

However, energy is also required on-farm and in meat processing for operating plant and machinery, refrigeration, heating, cooking, handling and for operating processing and auxiliary equipment. The majority of this energy is produced through burning petroleum based fossil fuels and through the consumption of electricity. Hence, direct (scope 1) and indirect (scope 2) emissions are produced from this energy consumption as a result of the various activities carried out in red meat production.

The principle greenhouse gas generated by the combustion of fossil fuels for energy is carbon dioxide. Fossil fuels are made up of hydrogen and carbon. When fossil fuels are burned, the carbon combines with oxygen to yield carbon dioxide. The amount of carbon dioxide produced depends on the carbon content of the fuel and the degree to which the fuel is fully combusted (i.e. the oxidation factor, which usually ranges between 98% and 99.5%); for example, for each unit of energy produced, natural gas emits about half and petroleum fuels about three-quarters of the carbon dioxide produced by coal (Department of Climate Change 2008a*)*. Small quantities of methane and nitrous oxide are also produced, depending on the actual combustion conditions. Methane may be generated when fuel is heated, but only partially burnt, and depends on combustion temperatures and the level of oxygen present. Nitrous oxide results from the reaction between nitrogen and oxygen in the combustion air (Department of Climate Change 2008c)*.*

The National Greenhouse and Energy Reporting (Measurement) Determination 2008 (Department of Climate Change 2008c) provides methods and criteria for calculating direct (scope 1) and indirect (scope 2) emissions from energy usage. Emissions from fossil fuels are most appropriately defined in terms of carbon dioxide generated per unit of energy because:

- carbon dioxide production is directly related to energy releases from a carbonaceous fuel
- energy content on a heating value basis is essentially independent of diluents such as ash, water and nitrogen.

There are four methods that allow for both direct emissions monitoring and the estimation of emissions through the tracking of observable, closely-related variables. Broadly, the four methods are as follows:

- Method 1 - the default methods, derived directly from the methods used for the National Greenhouse Accounts and the same as those used in OSCAR.
- Method 2—a facility-specific method using industry sampling and Australian or international standards listed in the Determination or equivalent for analysis.
- Method 3—a facility-specific method using Australian or international standards listed in the Determination or equivalent standards for both sampling and analysis of fuels and raw materials. Method 3 is very similar to method 2, but it requires reporters to comply with Australian or equivalent documentary standards for sampling.
- Method 4—direct monitoring of emission systems, on either a continuous or a periodic basis (Department of Climate Change 2008d).

For the grazing and lot feeding sectors of red meat production, method 1 will be the method used for estimating emissions. Method 1 specifies the use of designated emission factors and is most useful for emission sources where the source is relatively homogenous, such as from the combustion of standard liquid fossil fuels, where the emissions resulting from combustion will be very similar across most facilities (Department of Climate Change 2008d). For the processing sector, method 1 and other methods incorporating direct measurement of emissions may be employed.

Estimates of direct emissions (scope 1) from the combustion of individual solid, liquid or gaseous fuel types by method 1 are made by multiplying a (physical) quantity of fuel combusted by a fuel-specific energy content factor and a fuel-specific emission factor for each relevant greenhouse gas (in this case, carbon dioxide, methane and nitrous oxide). Emissions are estimated and expressed in tonnes of CO_2-equivalent (CO_2-e), which includes carbon dioxide as well as the global warming effect of the relatively small quantities of methane, nitrous oxides and perfluorocarbons.

Similarly, for estimates of indirect emissions (scope 2) from the purchase and consumption of electricity are made by multiplying physical quantity of electricity consumed by an electricity-specific emission factor. However if lifecycle emissions, such as emissions from extracting the fuel, and transmission losses, are to be incorporated, then Scope 3 emission factor as well as the Scope 2 factor are used. Emissions are estimated and expressed in tonnes of CO_2-equivalent (CO_2-e) per unit of electricity.

Fuel and electricity specific emission factors are outlined in the National Greenhouse Accounts (NGA) Factors (Department of Climate Change 2008e).

4.2 Energy Usage in the Grazing Sector

The major cattle growing areas in Australia are in rangelands (areas where domestic stock are grazed on native pasture) and semi-arid areas (pastoral zones). Similarly, the major sheep producing areas are in rangelands and semi-arid areas located in the southern sub-tropical and temperate zones.

Energy is required in the grazing sector to power generators, water supply, pumps, operate agricultural equipment (tractors, harvesters etc) and vehicles, stock handling and auxiliary equipment. The majority of this energy is produced through combustible petroleum based fossil fuels (diesel) and through the consumption of electricity.

Few studies have been undertaken to quantify the on-farm energy usage in the grazing sector of red meat production. Some data is available from farm surveys such as the Australian Bureau of Agricultural and Resource Economics (ABARE). Peters et al. (2009) undertook a labour-intensive collection of energy input data from two southern Australia red meat production chains as part of the LCI phase of the red meat production LCA. This data was collected form site inspections and on-farm surveys.

They reported that energy supply and use accounted for 5% of the total GWP contributions for sheep meat product. This equates to about 0.33 kg CO2-eq/kg HSCW. However, it is noted that these figures also include the use of energy in the preparation of feed for a low-density feedlot for finishing sheep.

For beef, on-farm energy supply and use accounted for less than 1% of the total GWP contributions and equates to about 0.077 kg CO2-eq/kg HSCW.

On-farm energy supply and use, as expected, is a relatively minor source of greenhouse emissions in the red meat supply chain. However, the Australian Governments proposed CPRS will affect grazing sector both directly (through costs associated with the need to either buy permits or reduce emissions) and indirectly through cost increases elsewhere in the economy.

Hence, adverse impacts from higher costs of inputs such as diesel, electricity, chemicals etc may be realised. As price-takers it will be very difficult for producers to pass on the additional costs to their customers and end users.

For the grazing sector, the critical strategies will be to firstly reduce consumption of energy and then to ensure that the energy that is consumed is produced from lower greenhouse intensive sources. Underpinning the first strategy will be improved identification and uptake of cost-effective energy efficiency opportunities. Davis et al. (2008) developed a framework for implementing a water and energy monitoring and evaluation system in the lot feeding sector which could easily be adapted to the grazing sector.

4.3 Energy Usage in the Lot Feeding Sector

Energy is fundamental to a beef feedlot production system. Energy is an important input cost for beef feedlots, and energy costs have risen significantly in recent years. Feedlots use petroleum-based fuels (primarily diesel) to operate vehicles, trucks, tractors and other mobile machinery for feed delivery, waste management and administration. Electrical energy is used to power grain handling and processing equipment, water supply and cattle processing equipment. Electrical energy is also used for lighting, heating, and cooling in offices and staff amenities. Natural gas, Liquefied Petroleum Gas (LPG- propane, LPG – butane) and solid fuel such as coal may also be used to generate steam for some methods of grain processing.

Despite this, there has been little research into energy use by beef feedlots. The energy requirements of feedlots have been estimated from several studies undertaken in North America in the 1970's and 1980's, however this data is now largely out of date.

The lot-feeding sector is under pressure from all levels of Government to report and reduce energy usage and GHG emissions.

Currently, the federal energy and greenhouse gas reporting obligations only apply at relatively high levels of energy usage (100 or 350 terajoules of energy for a single and corporation respectively, 25,000 tonnes of CO_2-e for a single facility or 87,500 tonnes of CO_2-e for a corporation). These thresholds will continue to reduce (Department of Climate Change, 2008b). Large, vertically integrated agricultural companies may meet these thresholds, resulting in reporting requirements for all subsidiary companies and feedlots in their control.

In Victoria, participation in the Environmental Resource Efficiency Plan (EREP) program is mandatory for industrial sites (beef feedlots fall into this category) that use more than 120 ML of water or 100TJ of Energy. If one threshold is tripped then water and energy usage is reported. The water threshold represents the water requirements for about 6,000-7,000 head-on-feed. There are other initiatives such as the National Pollutant Inventory (NPI) which could provide energy resource profiles.

Foreseeing these drivers for industry change, MLA has provided significant investment to quantify the energy usage within the lot feeding sector. This puts valuable information in the hands of the industry to improve resource efficiency, meet the requirements of legislation and improve the sustainability of the industry in the face of a variable climate.

Davis and Watts (2006) reported on-farm energy usage from nine cattle feedlots under a range of climatic, size and management conditions over the 2002 and 2004 years. They found that feed processing and delivery contributed on average between 45-80% of total energy usage depending on feed processing system. Significant amounts of energy may also be required for manure management (15%). Water supply (3%), administration (1%), cattle washing and cattle management contribute the remaining usage.

From this energy usage, scope 1 and scope 2 greenhouse gas emissions were estimated using method 1 outlined by the Australian Greenhouse office standard methodology (Australian Greenhouse Office, 2004). Emissions were standardised on a per-kilogram of HSCW gain whilst in the feedlot.

Further feedlot energy usage studies by Davis et al. (2008) have reported similar energy usage to those reported by Davis and Watts (2006).

GHG emissions from energy usage represent less than 1 kg CO_2-e per kg HSCW gain whilst in the feedlot. Feed processing energy usage contributes the highest emissions of around 0.1-0.5 kg CO_2-e per kg HSCW gain, depending upon type of feed processing system used (Davis and Watts, 2006). These emissions are based on the HSCW gain whilst in the feedlot.

Peters et al. (2009a) standardised feedlot energy use on a per-kilogram of HSCW produced in the red meat supply chain. Energy use in the feedlot contributes about 0.54 kg CO_2-e per kg HSCW or about 35% of total feedlot emissions (Figure 8). This represents about 5% of the total supply chain emissions (Peters et al. 2009a).

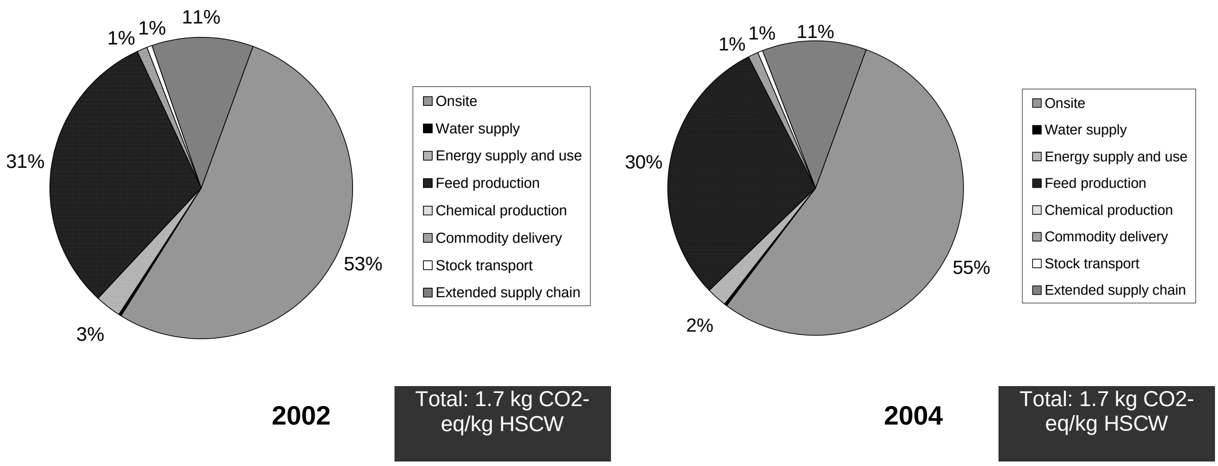

FIGURE 8: PERCENTAGE OF GHG EMISSIONS FROM OPERATIONS IN THE FEEDLOT (PETERS ET AL. 2009A).

Davis et al. (2008) have identified variability in energy usage between beef feedlots in the order of 100%. This suggests that opportunities exist for reducing energy usage and subsequently abatement of GHG emissions.

The majority of lamb finishing feedlots are small-scale opportunity enterprises and as such have feed processed off-site. Peters et al (2009a) collected energy use data for a sheep grazing and lamb-finishing operation in WA. The specific contribution by the feedlot operation is unknown, however combined grazing and finishing accounted for 5% of the total supply chain emissions.

Davis et al. (2008) have undertaken a comprehensive energy assessment of activities within the lot feeding sector as part of B.FLT.0339. The outcomes of this work have included benchmarking energy usage at an enterprise and developed a framework for implementing a water and energy monitoring and evaluation system. This framework will underpin improved identification and uptake of cost-effective energy efficiency opportunities.

4.4 Energy Usage in the Processing Sector

A considerable amount of work has been undertaken into the measurement and benchmarking of energy and water use in the processing sector of the red meat industry in Australia. This work has led to the production of an eco-efficiency manual by Meat and Livestock Australia (Meat and Livestock Australia Ltd 2002) for the industry. This manual documents the resource use and waste generation data for a typical meat processing plant in Australia, as illustrated in Figure 9. These inputs and outputs are quantified in Table 4.

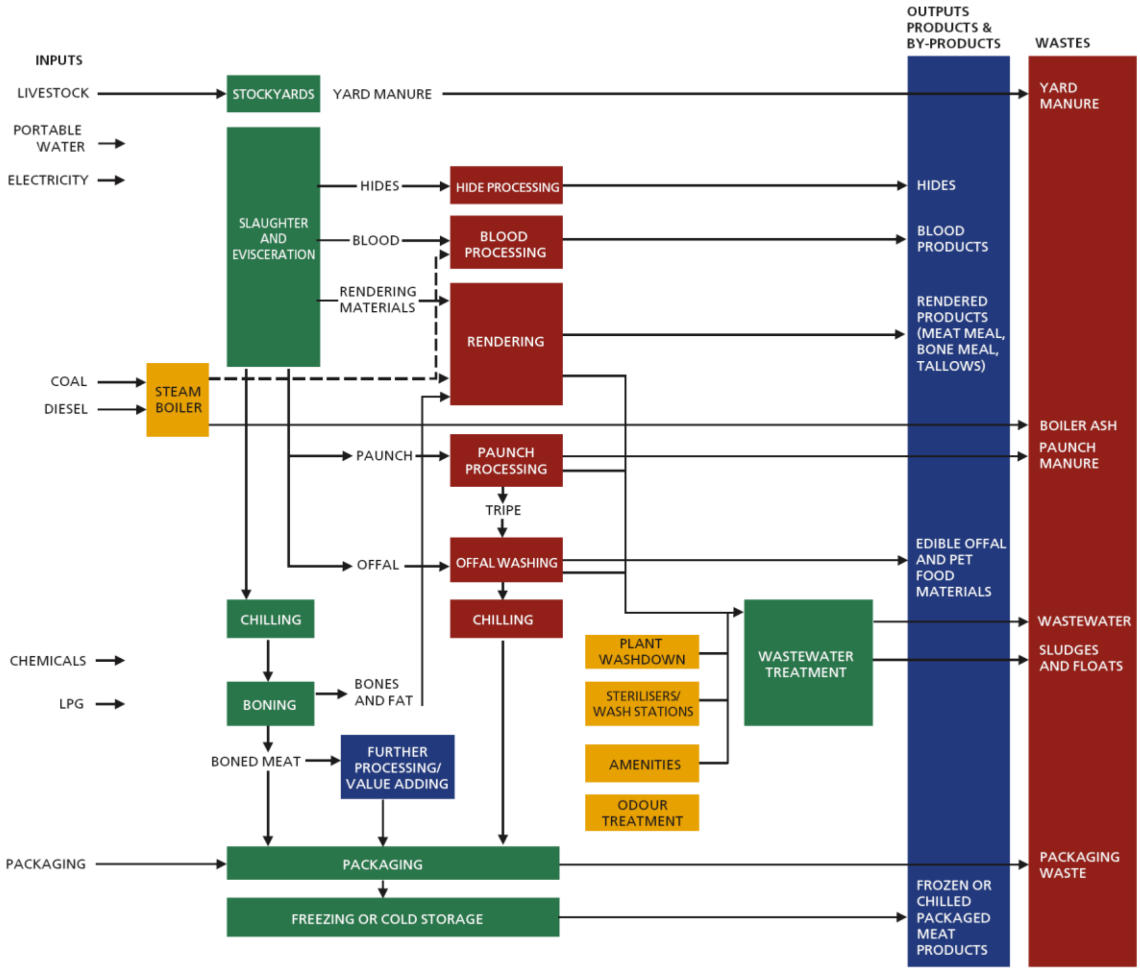

Figure 9: Process flow chart for a typical meat processing plant showing inputs and outputs (Meat and Livestock Australia Ltd 2002)

TABLE 4: RESOURCE USE AND WASTE GENERATION DATA FOR A TYPICAL MEAT PROCESSING PLANT (UNEP WORKING GROUP FOR CLEANER PRODUCTION, CITED IN MLA 2002)

Resources use		Daily quantity	Per unit of production
Water		1,000 kL/day	7 kL/t HSCW
Energy	Coal	8 t/day	53 kg/t HSCW
	LPG	113 m^3/day	0.8 m^3/t HSCW
	Electricity	60,000 kWh/day	400 kWh/t HSCW

The use of energy for both refrigeration and sterilisation is important at meat processing plants to ensure meat products are kept in good quality and food safety standards maintained, with storage temperatures in Australia specified by food safety regulations.

The amount of energy consumed at processing plants depends on a number of factors, including the age and size of the plant, the level of automation and the range of products manufactured (Meat and Livestock Australia Ltd 2007), with practices such as rendering consuming more energy.

Meat and Livestock Australia Ltd (2007) report total energy use in the processing sector in the range of 2 - 6 MJ/kg HSCW, with an average of 3.3 MJ/kg HSCW. This translates to greenhouse gas emissions of 0.25 - 0.90 kg CO_2-e/kg HSCW, with an average of 0.525 kg CO_2-e/kg HSCW.

Approximately 80–85% of total energy consumed by abattoirs is provided by thermal energy from the combustion of fuels (coal, fuel oil, natural gas and LPG (propane/butane) in on-site boilers. Thermal energy is used to heat water (both steam and hot water) for cleaning, rendering, blood coagulation and blood drying, with the remaining 15–20% of energy consumed provided by electricity. COWI Consulting (2000) provided a breakdown of thermal energy use in an abattoir, with rendering being over 40%, boiler losses 25% and hot water 14%.

The use of cleaner heat sources such as natural gas and liquefied petroleum gas, rather than coal and fuel oil is increasing due to environmental pressures to burn cleaner fuels, particularly those with lower sulphur contents that emit less sulphur dioxide (COWI Consulting 2000).

The electricity at meat processing plants is used for the operation of machinery, refrigeration, ventilation, lighting and the production of compressed air. Refrigeration is the largest user of electricity at meat processing plants. The other large usage areas are the multitude of motors that drive pumps, fans, conveyors, and hydraulic systems (Meat and Livestock Australia Ltd 2007). Table 5 provides an indicative breakdown of electricity use in an abattoir (COWI Consulting 2000).

TABLE 5: BREAKDOWN OF ELECTRICITY CONSUMPTION IN THE MEAT PROCESSING SECTOR (COWI CONSULTING 2000)

Purpose	Percentage of total
Refrigeration	59%
Boiler room	10%
By-products processing	9%
Slaughter area	6%
Compressed air	5%
Boning room	3%
Others	8%

Electricity consumption at meat processing plants is likely to represent the largest source of GHG emissions from energy use. Using the energy use data in Table 4 for coal, LPG and electricity, and GHG emission rates of 0.291, 0.069 and 0.0931 kg CO2-e/MJ for these energy sources respectively (Department of Climate Change 2008c), then GHG emissions for electricity consumption represents 73% of the total energy GHG emissions.

Refrigeration accounts for about 60% of the electricity consumption in meat processing. Refrigeration equipment uses refrigerants to facilitate the heat transfer process. Fluorocarbon refrigerants are synthetic chemicals which usually have a high global warming potential, and some still have the potential to cause damage to the ozone layer as well if released to the atmosphere. They were largely introduced as replacements for some ozone-depleting substances such as chlorofluorocarbons (CFCs).

Hence, refrigeration is a potential source of synthetic hydrofluorocarbons (HFC) GHG emissions. Synthetic GHG emissions account for around 1 per cent of Australia's emissions (or around one-fifth of industrial process emissions). These emissions are from the use of commercial and household equipment such as refrigeration, air-conditioning and high-voltage electrical equipment (Australian Greenhouse Office 2006a).

Only a small portion of the synthetic HFC which is contained in equipment will ever be emitted to the atmosphere. However, over the life of the equipment emissions occur as a result of slow, constant leakage, system failure resulting in complete venting of the gas to the atmosphere, and handling losses. When equipment reaches the end of its working life, a portion of the original gas may be available for recovery and destruction.

It is the organisation's responsibility to ensure that all refrigerant emissions are prevented where possible; failure to do so may lead to an offence under the *Ozone Protection and Synthetic Greenhouse Gas Management Act 1989* (Department of the Environment, Water, Heritage and the Arts 2009).

Department of Climate Change (2007a) provides a method for calculating CO_2-e emissions for a facility as the amount (stock) of HFC contained in each equipment type (see Table 6 for global warming potential), multiplied by a default leakage rate summed for each equipment type (see Table 7).

The amount (stock) of synthetic gas contained in an equipment type is estimated based on:

a) The stated capacity of the equipment according to the manufacturer's nameplate;
b) Estimates based on:
    i. The opening stock of gas in the equipment; and
    ii. Transfers into the facility from additions of gas from purchases of new equipment and replenishments; and
    iii. Transfers out of the facility from disposal of equipment or gas.

TABLE 6: GLOBAL WARMING POTENTIALS FOR HFCS (IPCC 1996)

Gas	Chemical Formula	Global Warming Potential
HFC-23	CHF_3	11,700
HFC-32	CH_2F_2	650
HFC-41	CH_3F	150
HFC-43-10mee	C_5H_2F10	1,300
HFC-125	C_2HF_5	2,800
HFC-134	$C_2H_2F_4(CHF_2CHF_2)$	1,000
HFC-134a	$C_2H2F_4(CH_2FC_3)$	1,300
HFC-143	$C_2H_3F_3(CHF_2CH_2F)$	300
HFC-143a	$C_2H_3F_3(CF_3CH_3)$	3,800
HFC-152a	$C_2H_4F_2(CH_3CHF_2)$	140
HFC-227-ea	C_3HF_7	2,900
HFC-236fa	C_3H2F_6	6,300
HFC-245ca	C_3H3F_5	560

TABLE 7: INDUSTRIAL PROCESSES EMISSION FACTORS AND ACTIVITY DATA FOR SYNTHETIC GASES (IPCC 2006)

Equipment type	Default annual loss rates - HFCs
Commercial air conditioning – chillers	0.09
Commercial refrigeration – supermarket systems	0.23
Industrial refrigeration including food processing and cold storage	0.16

Currently, HFC substances are not listed on Australia's National Pollutant Inventory. However, they are listed on international priority pollutant schemes such as the European Pollutant Emission Register Implementation (EPER 2009).

A number of strategies are available to both reducing energy usage at abattoirs and/or modifying energy sources to reduce GHG emissions. These are discussed further in the following section.

4.5 Mitigation Options

Efforts to mitigate GHG emissions from energy usage should focus on avoiding and abatement. Avoiding GHG emissions from existing facilities in the supply chain is the most cost effective approach and to some extent can be achieved by turning things off when not in use. For new facilities significant gains can be made through smart design and therefore, comparisons of the emission profile should be made when evaluating options for any new facility or process.

Abatement can take place in two basic ways; reducing energy use through energy efficiency projects, or generation or purchase of renewable energy. Improving steam generation and cogeneration with biogas are all good examples of abatement strategies.

The first step in evaluating mitigation options for energy usage within any process in red meat production is gaining an understanding of where the energy is being used within that process. This information allows areas for conservation to be identified and to determine where energy is being used inefficiently.

The following sections outline mitigation strategies for each of energy source. These strategies outlined can be divided into upstream (indirect), direct, and downstream (indirect) of the supply chain component. When considering mitigation strategies the actual mitigation practice may fall into a different category to the emission itself.

Electricity

Electrical energy is typically the greatest of all utility costs, despite the low unit cost, so significant savings are possible; it therefore makes economic sense to minimise energy consumption. The major mitigation strategies are outlined below.

Choose to purchase 'Green' electricity

Consumers can choose the source of their electricity through the purchase of 'green' electricity. Green electricity is sourced from sun, wind or hydro power, and is produced with minimal GHG emissions. The purchase of accredited renewable energy does not mean your electricity will come directly from a renewable source to your property, rather the equivalent amount of new renewable energy will be added to the electricity grid on behalf of the facility every year so you will be responsible for a reduction in greenhouse gas emissions.

Reducing demand for electricity

A number of strategies are available for reducing the demand for electricity. These include installation of energy efficient chilling systems, improving efficiency of electric motors and energy efficient lighting.

Energy use in a cold storage facility is affected by the amount of heat the refrigeration equipment must remove and the efficiency of the equipment. Refrigeration is the largest user of electricity at processing facilities and up to 10% of the power consumed can be due to heat ingress (Meat and Livestock Australia Ltd 2002). Therefore, power demand can be reduced by reducing heat ingress into refrigerated areas by ensuring doors are closed and use of plastic curtains.

Consideration of energy-efficient chilling systems such a plate chillers, turning off refrigeration at night are potential avenues to reduce power demand in facilities with these system. These are considered further in the meat processing eco-efficiency manual (Meat and Livestock Australia Ltd 2002).

Improving the efficiency of electric motors which power process equipment. This includes ensuring that motors are correctly sized for the function, modulate motor speed and reduce loads (e.g. pumping losses).

Installing an oversized motor can lead to unnecessary energy use. However, an oversized motor should not be replaced without making an accurate assessment on energy savings. An oversized motor could actually be just as efficient as a smaller sized motor.

Energy savings can be made by installing variable frequency drives (VFD) on motors which operate equipment at variable loads or that are oversized to cater for high contingent loads.

For facilities that use compressed air, high-efficiency compressors are one option for reducing power demand.

Different types of lights are available with different efficiencies and the use of task-level lighting is another strategy for reducing overall electricity usage.

Alternative sources of electricity

Photovoltaic systems and wind power are alternative energy options. These are most likely to be utilised for activities with existing diesel generators located in remote areas. Substantial savings in the fuel costs and reductions in CO_2 emissions could be realised with this renewable energy source.

A number of livestock industries are conscious of reducing their environmental impact, particularly through the production of waste by-products. Currently most by-products are utilised through land application or removed to landfill.

Internationally there has been a rapid increase in research and development in the area of conversion of livestock waste into energy, through various chemical, biochemical and thermal processes. Although overseas (particularly Europe and the United States) offers significant economic and legislative drivers to convert waste into energy that don't exist to the same extent in Australia, there may be opportunities to develop similar commercially viable operations in Australia or utilise currently operating plants to utilise by-products from red meat production.

The Australian Business Council for Sustainable Energy (2005) provides a comprehensive description of the primary and secondary methods to convert waste materials to energy or energy related products. In brief, drier forms of waste are usually converted through the thermal energy conversion paths, while wet wastes may be processed through biochemical pathways.

Meat processing and lot feeding produce by-products which are suitable for conversion into energy e.g. manure, wastewater, tallow. Abattoir waste is currently converted into energy at a number of meat processing plants (e.g. Rockdale Beef, Teys Bros, Kemp meats). Utilisation of by-products as a fuel source, offers these activities the potential to solve both disposal problems and provide an alternate fuel in different combustion technologies to reduce the dependence on fossil fuels and reduce greenhouse gas emissions. However, previous attempts to utilise animal waste as a sole fuel source have met with only limited success due to the higher ash, higher moisture, and inconsistent properties of the materials. The majority of work investigating solid and liquid wastes have been undertaken in the overseas in the US and Canada by researchers such as Annamalai et al. 2003a; Annamalai et al. 2003b; Sweeten et al. 2003.

Australian research (i.e. Lim and Headberry 2004; McGahan et al 2008; GHD Pty Ltd 2007) has focused on assessing the economic feasibility of biogas capture from uncovered effluent treatment lagoons – predominantly anaerobic lagoons and solid by-products from intensive livestock. This has included predicting waste production rates and the estimating the methane generation potential of these wastes.

Demand management

This practice involves creating the most efficient electric supply purchasing strategy, optimising load profiles, and reducing costs. At many facilities, the administrators are unaware of the rate structures of their electric bills. Hence, an energy management program should be put in place to ensure that electricity costs are kept to a minimum and identify efficiency options. There are multiple aspects of an energy management program including critical factors such as peak demand, power factor, and usage profile.

Diesel

Emissions from fuel (diesel) use are directly attributable to on-farm management practices and therefore can be controlled and minimised directly by the enterprise.

Reducing demand for diesel

If on-farm fuel is used for pumping, mechanisms available to reduce GHG emissions come from substituting with lower GHG emissions fuels such as solar or wind energy.

Alternative sources of energy

The level and type of air pollution generated by machinery depends largely upon the engine condition and the type of fuel used. CSIRO has evaluated GHG emissions from a range of alternative fuels (Beer et al. 2000). In relation to greenhouse gas emissions, renewable fuels such as bio-diesel (refined from vegetable oil) and ethanol were found to contribute least, while LPG and natural gas contribute significantly more. Various forms of diesel are the heaviest contributors. Good engine maintenance is important to ensure that whichever fuel is used, the lowest possible emission levels are achieved.

Improving diesel efficiency

This practice is aimed at best fuel efficiency and thus reduced fuel consumption for a range of mobile plant and equipment in use within the various activities of an enterprise. The key opportunities include:

- Match engine size to the task
- Maximise traction through load balancing and tyre settings
- Maintain the most efficient engine speed, according to the manufacturer's
- specifications
- Maintain machinery in good working order

Thermal Energy

Thermal energy, in the form of steam and hot water, is used for cleaning, heating water, sterilising and rendering predominantly in processing facilities but is also used during feed processing in feedlots. Steam and hot water is typically produced from boilers powered by coal, oil, gas or electricity.

Efficient steam generation and supply demand

Optimum operating efficiency of the boiler and ensuring that supply capacity is matched to facility demand is a key strategy to minimise fuel usage. There are various techniques available such as flue gas analysis which can help determine the operating efficiency of the boiler. A major variation in stack gas temperature indicates a drop in efficiency and the need for air-fuel ratio adjustment or boiler tube cleaning (Meat and Livestock Australia Ltd 2002).

Boiler efficiency can be improved by installing heat recovery equipment such as economisers or recuperators. An economiser is an air-to-liquid heat exchanger that recovers heat from flue gases to be used for other processes or pre-heat boiler feed-water. Fuel consumption can be reduced by approximately one percent for each 4.5ºC reduction in flue gas temperature. A boiler assessment audit can be undertaken to

Often steam supply capacity at the boiler house is too high compared to the plant’s steam demand, resulting in unnecessary fuel wastage. A plant’s steam demand is variable over a production day and over different months. Boilers must be run in a flexible manner to meet variable steam load. More metering instrumentation will help do this.

Other mitigation practices include minimising steam leaks to reduce steam wasted, so that less water and hence fuel is used to heat the additional water feed. Elimination of such leaks can save up to two percent of steam production costs (CADDET (2001) in Meat and Livestock Australia Ltd 2002).

Uninsulated steam and condensate return lines are a source of wasted heat energy. Insulation can help reduce heat loss by as much as 90%. Any surface over 50ºC should be insulated, including boiler surfaces; steam and condensate return piping and fittings. It is also important that sources of moisture are eliminated to prevent insulation from deteriorating and insulation that is damaged should be repaired (Meat and Livestock Australia Ltd 2002).

Reducing demand for steam such as reducing hot and warm water use in those facilities that heat water using steam will also result in reduced steam production costs.

Alternative sources of energy for steam generation

Coal, fuel oil, natural gas and LPG (propane/butane) are the typical sources of fuel used for steam generation. Natural gas (51.2 kg CO_2-e/GJ) and LPG (59.2 kg CO_2-e/GJ) are cleaner burning fuels compared with coal (88 kg CO_2-e/GJ) or fuel oils (72.9 kg CO_2-e/GJ). Therefore, the use of these fuels offers reduced emission of greenhouse gases and other air pollutants.

Facilities may not consider converting to natural gas because of the higher fuel cost. However in some situations, natural gas may be more economical overall due to lower labour, maintenance costs and avoided ash disposal costs (Meat and Livestock Australia Ltd 2002). Davis et al (2008) reported on significant savings in energy usage and costs realised by one feedlot in switching to LPG (butane) from LPG (propane).

The conversion from coal or fuel oil to natural gas would require the installation of a new boiler or substantial changes to the burner and fuel delivery system. Therefore the capital cost of this would make the conversion prohibitive for many facilities (Meat and Livestock Australia Ltd 2002).

In recent years, significant research has been undertaken on converting wastes to energy both within Australia and overseas. One of these techniques is the capture and utilisation of biogas from the anaerobic digestion of organic wastes. Processing plants and feedlots that use anaerobic lagoons for wastewater treatment will already be producing biogas, a methane-rich gas. These facilities also typically have steam generation requirements and the biogas can be used as a substitute for fossil fuels in boilers.

Refrigerants

Alternative Refrigerants

Alternatives to fluorocarbon chemicals exist that can help to mitigate some of the environmental risks. Often referred to as ‘natural’ refrigerants because the substances also occur in nature, these alternatives include ammonia, carbon dioxide and hydrocarbons. These substances have been used as refrigerants for many years; however, they are now finding their way into applications where previously fluorocarbons were the preferred option. Further information on ‘natural’ refrigerants can be found in Department of Environment and Water Resources (2007).

Maintenance Strategies

Around 50% of all leaks from commercial refrigeration systems occur at flared joints. Other likely sites are flexible hoses and damaged pipes.

Ensure equipment is appropriately charged with refrigerant, as overcharging can cause additional use of power and lead to higher losses during leakage. Undercharging will also cause excessive energy use.

Regular maintenance and checking of equipment to detect leakages, as these will also increase energy use.

4.6 Conclusions, Knowledge Gaps and Recommendations

4.6.1 Conclusions

It is most likely that emissions from livestock and manure management will be the key drivers of greenhouse emission mitigation in red meat production and not energy consumption as it contributes a relatively small component (5%) of the total supply chain emissions. However, reducing energy consumption will offer many sectors in red meat significant cost savings and as such will contribute towards reducing emissions.

Developed and developing technologies for converting organic by-products to energy are advancing rapidly with many new and proposed facilities operating or in the planning stages not only overseas but in Australia. This is the most strategic abatement option for energy consumption.

A factor inhibiting the adoption of energy efficiency and the use of renewable energy in Australia has been the low cost of energy and the lack of mechanisms to control demand. However, in recent years, the Australian Government and various State Governments have introduced incentives such as the Mandatory Renewable Energy Target (MRET), Renewable Energy Certificates (REC's) and NSW Greenhouse Abatement Scheme to drive projects such as biogas capture and use. These initiates along with the rising cost of carbon will also provide the impetus for converting organic by-products to energy.

The Australian Governments, *Carbon Pollution Reduction Scheme White Paper*, indicates that while agriculture will not be included in the CPRS before 2015, the ultimate inclusion of agriculture is desirable. The CPRS will affect red meat production both directly (through costs associated with the need to either buy permits or reduce emissions) and indirectly through cost increases elsewhere in the economy. Hence, even if agriculture is not included in the scheme it will still face adverse impacts from higher costs of inputs such as diesel, electricity, chemicals etc. As price-takers it will be very difficult for producers to pass on the additional costs to their customers and end users.

Therefore, the critical strategies will be to firstly reduce consumption of energy and then to ensure that the energy that is consumed is produced from lower greenhouse intensive sources. In this regard expanding the use of renewable energy that produces no greenhouse gas emissions is critical.

Hence, underpinning the first strategy will be improved identification and uptake of cost-effective energy efficiency opportunities. As the first step, this should include an enterprise level (farm or facility) energy audit (energy balance) to determine how efficiently energy is being used, identify energy and cost saving opportunities and highlight potential improvements in productivity and quality. Davis et al. (2008) have undertaken a comprehensive energy assessment of activities within the lot feeding sector as part of B.FLT.0339. The outcomes of this work have included benchmarking energy usage at an enterprise and sector level and developed a framework for implementing a water and energy monitoring and evaluation system. The mechanisms underpinning this framework and the methodology should be extended to the grazing sector.

Underpinning the second strategy will be waste-to-energy alternatives. Hence, the red meat industry must be provided with information on the feasibility of power substitution and heat generation using technologies and processes for converting organic by-products to energy.

4.6.2 Recommendations and Knowledge Gaps

The introduction of mandatory greenhouse and energy reporting (through the National Energy and Greenhouse Reporting Act – NGER) and the prospect of a national carbon pollution reduction scheme (CPRS) that places a monetary value on the right to emit GHGs poses a significant challenge to the red meat industry.

Hence, it is recommended that a program be implemented to clarify the current NGERS policies and methodologies from the perspective of the red meat industry. This program should:

- Increase awareness within the red meat industry of its obligations under current legislation and proposed legislation,
- Assist the respective industry sectors to meet their obligations under current legislation in measuring and reporting emissions,
- Develop tools to facilitate the abovementioned elements.

It is also recommended that a program be implemented to identify strategies to reduce demand and consumption of energy and to ensure that energy consumed is produced from lower greenhouse intensive sources. In this regard expanding the use of renewable energy is critical.

The potential for renewable energy sources is clear within the feedlot and meat processing sectors (energy generation from waste streams) and this needs to be promoted at an industry and government level.

5 GHG Emissions from the Processing Sector

Emissions from the processing sector arise primarily from energy usage and waste treatment. Energy usage has been covered in the previous section and is not expected to be significantly different for the processing sector; hence this has not been covered here. However, waste treatment emissions require some explanation. This section covers a brief review of the methodology for waste stream emission estimation of methane in the processing sector, and mitigation options.

5.1 Methane Emissions from Waste Management

The wastewater generated from meat processing plants is largely biological and contains very little material that is not fully degradable by biological means (CSIRO n.d.). Traditionally this waste is treated using a series of effluent treatment ponds (anaerobic to aerobic), after it has been pre-treated with some form of primary separation of solids. The main product produced during the anaerobic decomposition of organic matter is methane gas.

The Department of Climate Change methodology for estimating GHG emissions for waste (DCC 2007b) provides details of the production of methane from wastewater treatment.

> *Methane gas is the principal by-product of anaerobic decomposition of organic matter in wastewater. Large quantities of methane are not usually found in wastewater due to the fact that even small amounts of oxygen are toxic to the anaerobic bacteria that produce the methane. In wastewater treatment plants, however, there are a number of processes that foster the growth of these organisms by providing anaerobic conditions.*
>
> *Anaerobic conditions may also exist in malfunctioning aerobic treatment processes and collection systems. Significant amounts of methane emissions are attributable to wastewater treatment.*
>
> *As methane is generated by the decomposition of organic matter, the principal factor which determines the methane generation potential of wastewater is the amount of organic material in the wastewater stream. For industrial wastewater, Chemical Oxygen Demand (COD) is used. COD is a measure of the total material available for chemical oxidation (both biodegradable and non-biodegradable) (IPCC 2006).*

The methodology takes into account the amount emitted from on-site treatment (that not treated at municipal treatment plants) and the amount emitted from the disposal of the sludge left-over from treatment:

Emind = Emindnet + Emslind

Where: *Emindnet is the net mass of methane emitted from industrial wastewater that is not treated at municipal treatment plants.*

Emslind is the mass of methane emitted from sludge disposal generated by industrial wastewater treatment.

The methodology to determine the amount of methane generated from industrial wastewater in DCC (2007b) is given in IPCC (2000). Equation 6.B.1.2 is used to derive the methane emissions generated on a commodity by commodity basis:

$E_{mindgen} = ((Prod_i \times WW_{gen} \times COD_{gen}) – (COD_i \times F_{sl})) \times F_{an} \times EF_{mi}$

Where:

$E_{mindgen}$ is the quantity of methane generated from on-site industrial wastewater treatment

$Prod_i$ is the production of a commodity i

WW_{gen} is the wastewater generation rate from the production of commodity i – for meat processing the default is 13.71 m^3/t

COD_{gen} is the COD generation rate from commodity i - for meat processing the default value is 6.06 kg COD/m^3

COD_i is the COD load from the production/processing of commodity i (kg)

F_{sl} is the fraction of COD treated as sludge (in the absence of a plant-specific factor, the default value is 0.15, DCC (2008e)

F_{an} is the fraction of COD anaerobically treated - 0.43 is the default value for meat processing

EF_{mi} is equal to the IPCC default value of 0.25 kg CH_4/ kg COD (IPCC 2000).

Using the default values provided, the amount of COD in wastewater produced for each tonne of HSCW produced is 83.1 kg and the amount of COD removed as sludge is 12.5 kg per tonne of HSCW produced

Assuming the amount of sludge anaerobically treated is the same percentage as the amount of wastewater COD treated (43%) and that no sludge is removed from site for treatment, then 8.9 kg of CH_4 are produced per tonne of HSCW.

In order to calculate country specific data on COD production, the amount of wastewater produced and the concentration of COD of typical meat processing wastewater need to be obtained. Meat and Livestock Australia (Meat and Livestock Australia Ltd 2002) has previously published an eco-efficiency manual for meat processing in Australia. It documents the resource use and waste generation data for a typical meat processing plant in Australia, as illustrated previously in Figure 9. These inputs and outputs are quantified in Table 4 and show that COD production is approximately 38 kg/t HSCW produced.

TABLE 8: RESOURCE USE AND WASTE GENERATION DATA FOR A TYPICAL MEAT PROCESSING PLANT (UNEP WORKING GROUP FOR CLEANER PRODUCTION, CITED IN MLA 2002)

Waste generation		Daily quantity	Per unit of production
	Wastewater Volume	850 kL/day	6 kL/t HSCW
	Pollutant load		
	Organic matter (COD)	5,700 kg/day	38 kg/t HSCW
	Suspended solids	2,055 kg/day	13.7 kg/t HSCW
	Nitrogen	255 kg/day	1.7 kg/t HSCW
	Phosphorous	90 kg/day	0.6 kg/t HSCW
Solid waste	Paunch and yard manure	7 t/day	47 kg/t HSCW
	Sludges and floats	6 t/day	40 kg/t HSCW
	Boiler ash	0.7 t/day	5 kg/t HSCW

Using the Meat and Livestock Australia Ltd (2002) data and assuming the amount of sludge anaerobically treated is the same percentage as the amount of wastewater COD treated (43%) and that no sludge is removed from site for treatment, then 4.1 kg of CH_4 are produced per tonne of HSCW produced.

5.2 Nitrous Oxide Emissions from Waste Management

Wastewater produced from meat processing plants is also a source of nitrous oxide emissions. Nitrous oxide emissions have potential to occur both from the waste treatment process and during land application of effluent.

Department of Climate Change (2008e) provides no method for calculating nitrous oxide emissions from meat processing wastewater treatment. However, the Department of Climate Change (2007) does provide estimates of nitrous oxide emission rates for various swine manure management systems based upon the fraction of nitrogen in the waste, with anaerobic lagoons estimated to emit 0.1% of the nitrogen added. Department of Climate Change (2008e) also provides no method for calculating nitrous oxide emissions from the irrigation of meat processing wastewater. However, the Department of Climate Change (2007) does provide an estimate of the nitrous oxide emissions for animal manures applied to land of 1% of applied nitrogen. Table 15 of the guideline states that nitrous oxide emissions for organic fertiliser range from 0.21 – 3.31% of the applied nitrogen.

A more detailed assessment of nitrous oxide emissions from waste management and wastewater irrigation, including a review of the methodologies applied, are provided in a separate report as part of this project. Considering the lack of a specific methodology for the processing sector and the similarity of the underlying processes involved in emissions from feedlot effluent, these sections may be viewed as interchangeable. However, accounting for nitrous oxide in the processing sector is clearly an issue that requires consideration if overall emissions are to be reduced.

5.3 Mitigation Options

Meat processing plants generate large amounts of organic waste as mentioned previously, primarily from anaerobic ponds. Other organic waste streams at abattoirs include solid wastes in the form of paunch and holding yard waste.

The anaerobic treatment process in the treatment ponds produces biogas that is made up mainly of methane (typically 60 – 80 % by volume). The biogas also contains carbon dioxide and smaller amounts of other components, such as hydrogen sulphide (H_2S), nitrogen and oxygen.

The main mitigation option for the meat processing sector will be methane capture and utilisation. This can be done by either; i) covering ponds with impermeable (high density polyethylene – HDPE) covers and flaring methane (low cost), or ii) building purpose built digesters to generate methane that can be used for heat or electricity generation (high cost). Captured biogas with a typical methane content of 65% has a heating value of 22.4 MJ/m^3, compared with pure methane (natural gas), with a heating value of around 40 MJ/m^3 (Meat and Livestock Australia Ltd 2002). The captured biogas generally needs to be treated to remove impurities (moisture and H_2S) before being used for heating of boilers or converted to electricity to offset power consumption. Co-generation can also be used to capture excess heat from electricity generation. A series of feasibility studies on biogas utilisation by the UNEP Working Group for Cleaner Production 1999 (cited in MLA 2002) found that the energy available from biogas from the digestion of food processing wastewaters typically provide 10-20% of a plant's thermal energy requirements.

Reducing the quantity or organic matter in the meat processing waste stream will also reduce the amount of methane generated due to lower the amount of COD to be anaerobically treated. This can be achieved through the further removal of solid material before wastewater. Ways of reducing organic load include; screening of individual wastewater streams to recover lost product, segregation of hot water streams to improve fat recovery and removal of stick-water solids using evaporation. These options are discussed in further detail in the Eco-efficiency manual for meat processing (Meat and Livestock Australia Ltd 2002).

Another area requiring further investigation is the use of either naturally occurring or artificial surface crusts on anaerobic treatment ponds. The theory is that surface crusts can reduce emissions by providing an environment for the bacterial oxidation of CH_4 (Petersen et al. 2005; Sommer et al. 2000). These methane oxidising bacteria, known as methanotrophs, are naturally occurring in the environment (i.e. in rice fields, sediments and landfill covers). Anaerobic treatment ponds at meat processing plants often form a natural crust due to the fat content and fibrous nature of the waste steam entering the pond. The formation of these crusts is often encouraged to reduce odour emissions. The work by Petersen et al. (2005) reported that surface crusts formed on both cattle and pig slurry could significantly reduce CH_4 emissions. There appears to be little work done on the effect of these surface crusts on reducing methane emissions from meat processing ponds however. This may be an important mitigation option for smaller processing plants if pond covers prove to be too expensive, or may be a transition approach to reducing emissions.

5.4 Conclusions, Knowledge Gaps and Recommendations

5.4.1 Conclusions

The capture of methane gas from meat processing wastewater offers the potential to both reduce GHG emissions and reduce the reliance of energy from non-renewable resources. Advancements in this area are progressing rapidly both in Australia and overseas, with many industries (including meat processing) investigating and adopting these technologies.

However, as discussed in section 4, a factor that is inhibiting this type of technology is the low cost of energy which does not incentivise further development in this area. This is now changing with various Australian Government and State Government incentives such as the MRET, REC's and NSW Greenhouse Abatement Scheme to increase the uptake of biogas capture and use.

Providing the processing sector with information on the feasibility of power substitution and heat generation using technologies and processes for converting organic by-products to energy will likely further increase uptake.

5.4.2 Knowledge Gaps and Recommendations

It is recommend that a program be implemented to clarify the current government GHG methodologies (the DCC manual) and policies (i.e. the NGERS) from the perspective of the processing sector. This program should include;

- Fundamental research to characterise GHG emissions from meat processing waste streams under a variety of current and potential waste treatment strategies,
- Development of tier 3 methods for alternative waste management strategies that may be used by the Australian meat processing sector where significant differences in emission levels are identified,
- Pathways to increase awareness within the processing sector of its obligations under current legislation and proposed legislation.
- Assisting the processing sector to meet their obligations under current legislation in measuring emissions and reporting emissions.
- Development of tools to facilitate the abovementioned elements.

It is also recommended that a program be implemented to identify strategies to reduce demand and consumption of energy and to ensure that the energy that is consumed is produced from renewable energy as much as possible.

6 Vegetation Management in the Red Meat Industry

6.1 Introduction

Australia's grazing industries are responsible for the management of some 47% of Australia's land area (ABS 2009). It follows that, as land managers, graziers are the most significant group of vegetation managers in the country. This role is perhaps the most important that the red meat industries hold with respect to carbon management and is also the least acknowledged. It goes to follow that vegetation (and soil management) may offer the greatest opportunities for the industry to capitalise on carbon management if the right combination of policy and scientific research are brought together. Two factors that stand against recognition of this role are the current state of vegetation legislation (that have already begun the process of taking ownership of vegetation on private land) and the NGGI accounting structure, which divides vegetation management sequestration from other land uses that may occur on the same property (such as livestock emissions).

Considering the rapid movement in government policy in this area, policy implications are perhaps even more important that the underlying science involved, though for a holistic industry approach to be developed a considerable basis of understanding will be required in a short amount of time.

It is well understood by the grazing industries that healthy, productive and diverse ecosystems are important to the viability and sustainability of grazing properties and the industry as a whole. The native grasses, shrubs and trees found in grazing systems in Australia are not only suitable for grazing, they also provide habitats for a diverse range of native animal, insect, bird and plant life. The balance between trees and grass is a major factor influencing productivity, environmental stability and biodiversity (Meat and Livestock Australia Ltd 2008).

The land use management sector, including the inter-related factors involved in vegetation management, vegetation sinks, land clearing and soil carbon continue to come under close scrutiny because emissions related to land use are crucial to the establishment of Australia's baseline for the Kyoto Protocol (emissions dated from 1990), and because sinks will be a vital emissions offset mechanism into the future (UNFCCC 2007). Hence, State, Territory and the Commonwealth Governments have introduced a range of legislation, regulations and policies that will maintain and increase carbon storage from vegetation on private land.

Domestically, stricter land clearing regulations have been identified as a key area for cost effective emissions reduction, and the development of an appropriate regulatory framework has been foreshadowed by the Australian Government according to McKinsey (2008). The Stern Review has made clear that "curbing deforestation is a highly cost-effective way of reducing GHG emissions and has the potential to offer significant reductions fairly quickly" (The Stern Review 2006). Garnaut is also of the view that "the level of greenhouse gases emitted and sequestered by land-use change, agriculture and forestry will be critical in determining the difficulty and cost of eventually stabilising greenhouse gas presence in the atmosphere" (Garnaut 2007). *Essentially these perspectives underscore the moves by government towards taking ownership of vegetation (and therefore vegetation management) on private land for the purposes of offsetting national emissions, particularly emissions from other growth sectors.*

Deforestation is the most carbon intensive operation covered by "Land Use Change", and refers to the deliberate, human induced removal of forest cover and replacement with pasture, crops or other uses (UNFCCC 2007). Emissions from deforestation are included in national emissions accounts under the Kyoto rules. Emissions from deforestation are the result of the burning of removed forest cover, the decay of unburnt vegetation, and emissions from soil disturbed in the process of land clearing. This is offset to some extent by carbon sequestration due to regrowth of vegetation on previously cleared land. Estimates of emissions from Land Use Change depend on the area of forest cover removal and the method of forest conversion and land development. They rely on estimates of the amount of carbon sequestered in biomass and soils, which differ by vegetation type, geography and climate. Land clearing rates in Australia are influenced by factors such as market forces, technology change, climatic events, e.g. drought, and government policy.

Under Article 3.3 of the Kyoto Protocol, it is mandatory for Australia to report on the following activities if they were commenced in or after 1990 (UNFCCC 2007):

- **Afforestation** — defined as "the direct human induced conversion of land that has not been forested for a period of at least 50 years to forested land through planting, seeding and/or the human-induced promotion of natural seed sources."

- **Reforestation** — defined as "the direct human induced conversion of non forested land to forested land through planting, seeding and/ or the human-induced promotion of natural seed sources, on land that was forested but that has been converted to non forested land. For the first commitment period of 2008-2012, reforestation activities will be limited to reforestation occurring on those lands that did not contain forest on 31st December 1989."

- **Deforestation** (land use change or land clearing) — defined as "the deliberate human induced removal of forest cover and replacement with pasture, crops or other uses."

Land use change sector emissions in 2007 represented 6 % of the Australian total, and at 76.8 Mt CO_2-e were 44 % lower than the 1990 emissions of 136.5 Mt CO_2-e (Department of Climate Change 2009b). Land use change emissions have declined significantly over the period since 1990 as shown in Figure 10. These reductions have resulted from factors such as commodity price fluctuations, climatic events *and the introduction by state and territory governments of new land clearing regulations*. Over the period 1995 to 2005, Queensland and NSW contributed 68% and 17% respectively of average national Land Use Change emissions. Bans on broad scale land clearing in Queensland and significant tightening of clearing controls in NSW have contributed strongly to this reduction.

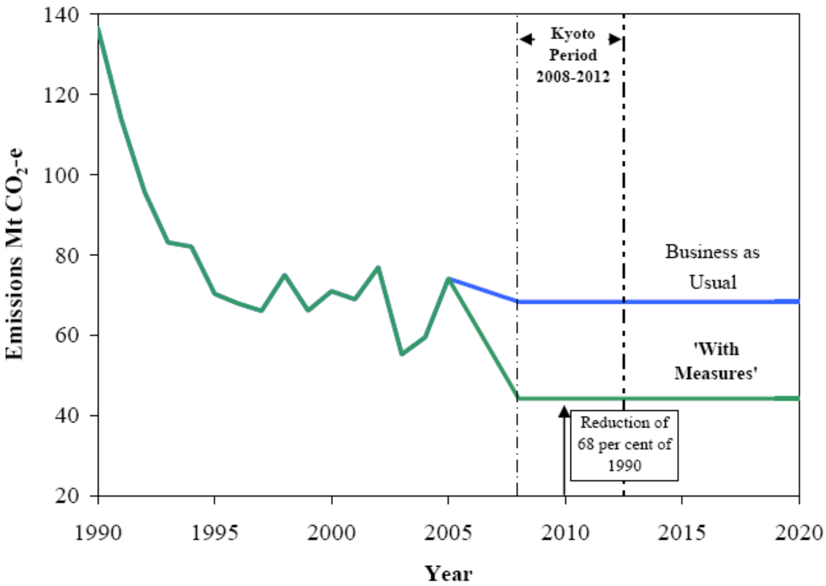

FIGURE 10: GHG EMISSIONS FROM LAND USE CHANGE 1990-2020 (DEPARTMENT OF CLIMATE CHANGE 2009B)

Further, because 1990, the year on which emissions targets under the Kyoto Protocol were based, was an atypically high year for Australia's emissions from land clearing, subsequent restrictions on land clearing in Queensland and NSW underpinned the fact that Australia's total greenhouse gas emissions hardly increased at all between 1990 and 2000. Indeed, this is the major reason for Australia being on track to meet its Kyoto Protocol emissions target in the period 2008-2012, particularly given the strong emissions growth from almost all other sectors of the economy (Australian Greenhouse Office 2006c).

6.2 Legislation

The State, Territory and the Commonwealth Governments legislation, regulations and policies to help maintain and enhance native vegetation and biodiversity can be accessed on the internet as shown in Table 9. A brief overview of the State, Territory and Commonwealth legislation is presented in the following sections and Table 10.

TABLE 9: WEBSITES FOR KEY NATIVE VEGETATION AND BIODIVERSITY LEGISLATION IN EACH STATE AND TERRITORY

State	Website
ACT	http://www.legislation.act.gov.au/
NSW	http://www.legislation.nsw.gov.au and http://www.environment.nsw.gov.au
NT	http://www.dcm.nt.gov.au/
QLD	http://www.legislation.qld.gov.au and http://www.nrw.qld.gov.au
SA	http://www.legislation.sa.gov.au/
VIC	http://www.legislation.vic.gov.au/ and www.dse.vic.gov.au
TAS	http://www.thelaw.tas.gov.au/
WA	http://www.slp.wa.gov.au/ and http://www.dec.wa.gov.au

6.2.1 Commonwealth Legislation

The Federal government's central piece of environmental legislation is the Environmental Protection and Biodiversity Conservation Act 1999 (EBBC Act). The Act provides a legal framework to protect and manage matters of national environmental significance including: world heritage sites, national heritage places, wetlands of international importance, nationally threatened species and ecological communities, migratory species, commonwealth marine areas and nuclear actions. Additionally, the Act confers jurisdictions over actions that have a significant environmental impact on commonwealth land or that are carried out by a Commonwealth agency. The EPBC Act effects any group or individual including farmers and landowners when a proposal for a project has the potential to have a significant impact on a matter of national environmental significance. The Department of the Environment, Water, Heritage and the Arts release the referred proposal to the public for comment on whether the project is likely to have a significant impact on a matter of national environmental significance. Public comments are reviewed, and the minister or delegate makes the final assessment on the project (Department of the Environment, Water, Heritage and the Arts, 2009).

6.2.2 State Legislation

In addition to commonwealth legislation, each state and territory has their own legislation regarding vegetation management including land clearing. Legislation is subject to change and producers are recommended to research current federal, state and local government legislation before undertaking any projects, which are associated with clearing or land use changes. Refer to Table 9 for a summary of state legislation that may apply to red meat producers.

TABLE 10: OVERVIEW OF STATE LEGISLATION RELATING TO NATIVE VEGETATION MANAGEMENT

Jurisdiction	Legislation	Description
Australian Government	Environment Protection and Biodiversity Act 1999 (EPBC Act)	Provides a legal framework to protect and manage matters of national environmental significance including: • World heritage sites • National heritage sites • Wetlands of international importance • Nationally threatened spp. and ecological communities • Migratory species Confers jurisdiction over actions that have a significant environmental impact on Commonwealth land or are carried out by a Commonwealth agency.
ACT	Nature Conservation Act 1980	
NSW	Native Vegetation Act 2003	Aims to end broad scale clearing across the state. All clearing requires approval through either a Property Vegetation Plan (PVP) or a Development Consent unless it is: i) on land that is excluded from the NV Act ii) categorised as excluded clearing iii) a permitted clearing activity Permitted clearing activities include: • construction of a single dwelling • routine agricultural management activities (RAMAs) • clearing of non-protected growth • sustainable grazing • clearing of certain groundcover • to continue existing farming activities.
	Threatened Species Conservation Act 1995	Provides a number of tools for conserving biodiversity and protecting threatened species, populations, communities and their habitats at the landscape scale. Approved PVP and RAMAs do not require a separate threatened species licence.
	Fisheries Management Act 1994 no 38	Degradation of native riparian vegetarian along water courses.
NT	Territory Parks and Wildlife Conservation Act 2000	Makes provisions for and in relation to the establishment of parks and reserves and the study, protection, conservation and sustainable utilisation of wildlife.
	Environmental Assessment Act (1994)	Establishes a framework for the assessment of potential or anticipated environmental impacts of development.

QLD	Vegetation Management Act 1999	Regulates clearing of remnant vegetation and some non-remnant vegetation on freehold land and state tenures. • Permits are not required to conduct a native forest practice on freehold land • Clearing applications are required for legitimate clearing purposes. For example a development is required for fodder harvesting.
	Vegetation Management (Regrowth Clearing Moratorium) Act 2009	From 8 April 2009 there is a moratorium in place for clearing vegetation within 50 m of a watercourse in a priority reef catchment and for clearing endangered regrowth vegetation in rural areas on free-hold and agricultural and grazing leasehold land.
	Native Conservation Act 1992	Aims to protect nature. Allows for the dedication and declaration of protected areas and the management of protected areas.
SA	Native Vegetation Act 1991	Aims to provide incentives and assistance to landowners in relation to the preservation and enhancement of native vegetation. Controls the clearance of native vegetation
VIC	Flora and Fauna Guarantee Act	Aims to protect threatened species or communities. A Protected Flora Licence or Permit is required to collect protected native plants or carry out works on **public** land which may kill, injure or disturb protected native plants. Not required for works on **private** land.
	Planning and Environment Act 1987	Sets up a system of planning schemes to regulated land use in VIC. • **Native vegetation planning permit** from the local council is required to remove, destroy or lop vegetation unless exemptions apply. • **Property Vegetation Plan (PVP)** is a voluntary agreement between a landholder and the Department of Sustainability, which considers how all vegetation on the property will be managed in the next 10 years.
	Catchment and Land Protection Act 1994	The Act imposes various duties on land owners to take all reasonable steps to: • avoid causing or contributing to land degradation which causes or may cause damage to another landowner; • conserve soil, • protect water resources

TAS	Forest Practices Act 1985 Forest Practices Regulations 2007 Nature Conservation Act 2002	This Act regulates forestry on both public and private land in Tasmania and is administered by the Forest Practices Authority. Controls apply to the clearing, removal or destruction of native trees or other woody plants, including any seedlings, that have the potential to grow to a height of >5 m. Controls apply the clearing and conversion of *threatened* non-vegetation There are no controls on clearing non-forest vegetation that is not threatened. A certified forest practices plan is required to authorise controlled land clearing unless an exemption is provided in the Forest Practices Regulations. Details exemptions from the requirement to have a forest practices plan to authorise clearing. Establishes provisions for the conservation and protection of the fauna, flora and geological diversity of TAS and provides for the declaration of national parks and other reserved land and for related purposes. Affected landowners may be eligible for compensation.
WA	Environmental Protection Act 1986	Clearing native vegetation is prohibited unless clearing permit is granted by DEC or clearing is for an exempt purpose including: • Clearing to construct a building • Clearing to collect firewood • Clearing to obtain fencing or farm materials • clearing for fence lines • Clearing for vehicular and waling tracks • Clearing isolated trees The total area cleared in one financial year must not exceed 1 ha. Exemptions don’t apply to environmentally sensitive areas and additional controls are placed on country areas water supply catchments (Department of Environment 2005b)

Note: This table does not constitute a complete register of relevant legislation, relating to vegetation management at a property level. While key Acts have been identified, additional requirements from other legislation are not listed here.

6.3 Offsetting emissions from the production of red meat with vegetation sinks

The science is both strong and coherent in accurately assessing long-term gains and losses of carbon, and other emissions, from the forestry and land use sector. For decades landholders and government agencies have been measuring and monitoring forest status and growth using a combination of techniques including direct field measurements, satellite and aerial photography and computer modelling. Many protocols for measuring and monitoring carbon project benefits already exist (Department of Climate Change 2007).

Although land clearing has been reduced in recent years, it remains an important source of greenhouse gas emissions in Australia. Planting trees, as opposed to clearing them, has the opposite effect on greenhouse gas emissions. On a national basis, or at least in most states, forestry promises to provide significant opportunities for sequestering carbon in so-called ‘sinks’ and thereby reducing the net cost of emissions abatement.

There are major opportunities to reduce emissions and increase carbon sequestration in land use sectors. This will have implications on the northern Australian beef industry, a significant component of Australia’s red meat industry where land areas a still relatively undeveloped. As industry and agriculture develops further and the population continues to grow, there will be significant commercial pressure to clear more land.

Garnaut (2008) estimated that about half of Australia's annual emissions could be absorbed by improved land management practices that enabled revegetation in Australia’s arid and semi-arid rangelands. However it has not been shown how this can be done without significantly reducing the agricultural productivity of these areas.

Methodology for Estimating Land Use Change Emissions

Accounting capability for Australia’s land based sectors has been developed through the National Carbon Accounting System (NCAS) (Department of Climate Change 2005). The NCAS is being progressively developed to provide a complete greenhouse gas accounting capability for agriculture, forestry and land use change (including all carbon pools, gases, lands and land use activities). The eventual capacity will be a full spatial enumeration with emissions and removals calculated using a process-based, mass balance, carbon and nitrogen cycling ecosystem model.

Emissions from Land Use, Land Use Change and Forestry (LULUCF) are provided in the National Greenhouse Gas Inventory (NGGI). The GHG emissions associated with LULUCF are estimated using the National Carbon Accounting System (NCAS). GHG emissions are calculated with the most recent available data using Kyoto Protocol accounting rules, *IPCC 1996 Guidelines for National Greenhouse Gas Inventories* (IPCC 1997) and the *IPCC Good Practice Guidelines for Land Use, Land Change and Forestry* (IPCC 2003), whilst taking into account Australian conditions.
The GHG emissions associated with vegetation are divided into the following categories:

- Harvested Native Forests for *Forest land remaining Forest land;*
- Forest Plantations for *Forest land remaining Forest land* and *Grassland converted to Forest land;*
- *Forest land converted to Grassland* and *Cropland;*
- *Harvested Wood Products.*

The NCAS has been under development to provide a complete (all carbon pools, gases, lands and land use activities) GHG accounting capability for agriculture, forestry and land use activities by 2008. Management practices, vegetation changes and climate variability are the principle causes of GHG emissions associated with vegetation (land use, land use change and sinks) and the sources of annual variability. Modelling is based on the use of medium resolution (50 m and 25 m) Landsat satellite data in a time-series since 1972. The medium resolution data is used to determine changes in forest and sparse woody vegetation to determine plantation areas, ages and types. Land use is mapped using a coarse 1 km resolution (NOAA) satellite data in 16-day time series, constrained to agricultural survey statistics. Monthly climate maps are generated from 1 km resolution satellite data to estimate the annual variation in emissions due to climatic process drivers. Additionally, the model uses data from Australia's national forest inventory and other independent sources and checked with field measurements including forest growth rates and soil carbon.

The land use change and forestry sector emissions reported in the NGGI do not include N_2O emissions from nitrogen fertiliser application and disturbance associated with land conversion. These emissions are reported in the Agricultural sector.

Carbon dioxide fluxes are the main GHG involved in the land use change and forestry sector. This sector continues to come under close scrutiny as 1990 land clearing emissions are crucial to the establishment of Australia's baseline for the Kyoto Protocol and because sinks will be a vital emissions offset mechanism. There has been considerable research into the greenhouse inventory methodology for clearing and plantation sinks, while relatively little has been done in relation to the methodology for carbon storage in agricultural systems.

The land management sector of the red meat industry has the capacity to sequester a large amount of atmospheric carbon and contribute to GHG emissions abatement. Hence, it is in the long-term interests of the red meat industry that it investigates and implements effective practices to reduce its GHG emissions and enhance the amount of carbon it sequesters. The capacity for carbon sequestration in rangeland areas has been reviewed by Garnaut (2008) and more recently for Queensland by Gifford & McIvor (2009). Gifford & McIvor (2009) acknowledge the considerable difficulties in making these estimates and concede that livestock numbers may need to be reduced to achieve increased carbon sequestration in some instances, particularly on land classified as 'degraded'. Interestingly, Gifford & McIvor (2009) suggest that less of Australia's rangelands are degraded than previously reported by Garnaut (2008). Following a review of these reports there appears to be a lack of attention given to win-win options where carbon sequestration can occur without compromising livestock productivity, though this is obviously the best solution for the livestock industries and the economy if it can be achieved. Consequently such projections should be viewed with caution by red meat producers. None-the-less, carbon sequestration in vegetation may be a valuable option for livestock producers if carbon is a tradable commodity.

6.3.1 Carbon Sequestration

Plants take up (sequester) carbon dioxide (CO_2) from the atmosphere as they are growing, through the process of photosynthesis. During photosynthesis, carbon is stored in plant biomass, releasing oxygen back into the atmosphere. In a forest, carbon is stored above ground in plant tissue, in litter and debris on the forest floor, and below ground in plant roots and the soil. When trees shed leaves, twigs, branches, bark or roots, the stored carbon returns to the atmosphere or enters the soil through using decaying agents (fungi, microorganisms and insects). A forest system is a sink when the forest is actively growing and sequestering carbon at a rate that outweighs any soil carbon emissions (Department of Environment and Heritage 2006).

Acceptable forms of tradable carbon from vegetation

Australia will be implementing the rules outlined by the Kyoto Protocol in regards to the establishment and design of forests for carbon sequestration. Under the protocol, parties can only count increases in forest carbon over Australia's commitment period (2008-2012), from forests established after 1 January 1990 on previously cleared land. This is because reforestation sequestration and emissions (i.e. from harvesting, pests or fire) are relatively well understood, and reliable, cost effective methods for estimating these are readily available. Plantation or native forests established prior to 1990 are currently not included in the carbon pollution reduction scheme (Department of Climate Change 2008). It is also not clear yet as to which forest components will be included in the trading scheme (i.e. carbon from upper storey (trees) or lower storey (shrubs) above ground biomass, root biomass, woody and other debris or soil carbon).

Forest Definitions

A forest is defined under the Kyoto Protocol as "a minimum area of land of 0.05 -1 ha with tree crown cover (or equivalent stocking level) of more than 10-30 %. Trees must have the potential to reach a minimum height of 2-5 m at maturity in situ. A forest may consist either of closed forest formations (multiple storeys of vegetation cover) or open forest. Young natural stands and all plantations which have yet to reach a crown density of 10-30 % or tree height of 2-5 m are included under forest, as are areas normally forming part of the forest area which are temporarily unstocked as a result of human intervention such as harvesting or natural causes, but which are expected to revert to forest (Department of Environment and Heritage 2006). See 6.1 for a definition of reforestation under the Kyoto Protocol.

The Australian definition of a forest for Kyoto Protocol accounting purposes has the following criteria:

- a minimum area of 0.2 hectares (for reasons of detection),
- at least of 20 % tree crown cover, and
- a tree height of two metres.

Carbon trading and Sequestration

Under the proposed Carbon Pollution Reduction Scheme White Paper (Department of Climate Change 2008b), forest entities can receive permits for the net GHG sequestration during the growing phase of a forest that would count towards Australia's international commitments. Permits will be issued on a stand-by-stand basis for the projected GHG removals, after scheme commencement, up to a permit limit set by the forest regulator (set below expected removals). Forest entities would have to surrender permits for any net emissions that lead to a change in the permit limit (e.g. land converted to a non-forest use, or if a harvested forest is not replanted). Net carbon sequestration and emissions would be calculated using a standard methodology such as the National Carbon Accounting Toolbox (Australian Greenhouse Office, 2005). Permits will be issued on an average-crediting basis (i.e. permits will generally not need to be surrendered on harvest or following fire and then re-issued when the forest is re-established). The permit limit is then reduced to create a 'risk reversal buffer' removing the need to surrender permits in the event of natural disturbances (e.g. fire, insect damage). Figure 11 shows an example of C sequestration in a forest grown for non-harvest purpose and Figure 12 the C sequestration from a harvested forest re-established over time.

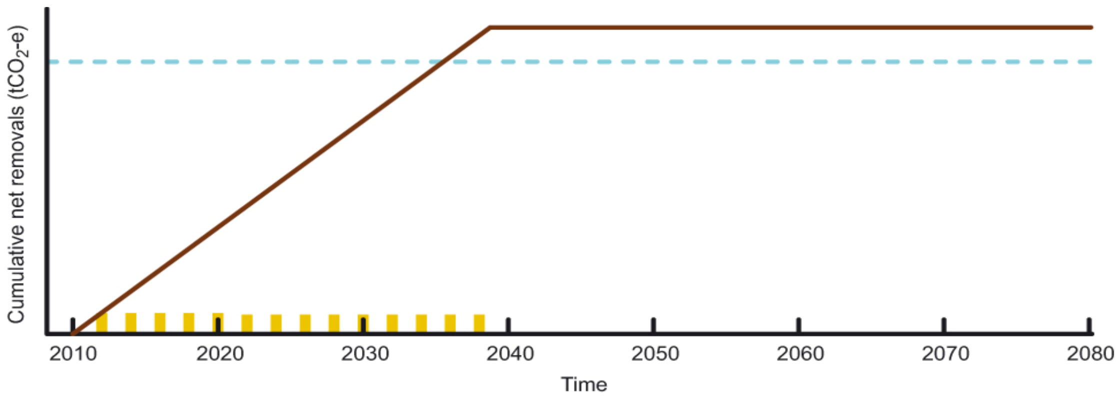

FIGURE 11: EXAMPLE FOREST GROWN FOR NON - HARVEST PURPOSES (SOURCED FROM: DEPARTMENT OF CLIMATE CHANGE 2008B)

*The blue-dashed line represents the permit limit including the 'risk of reversal buffer' and the yellow bars indicate the permits issued during the forests growing stages.

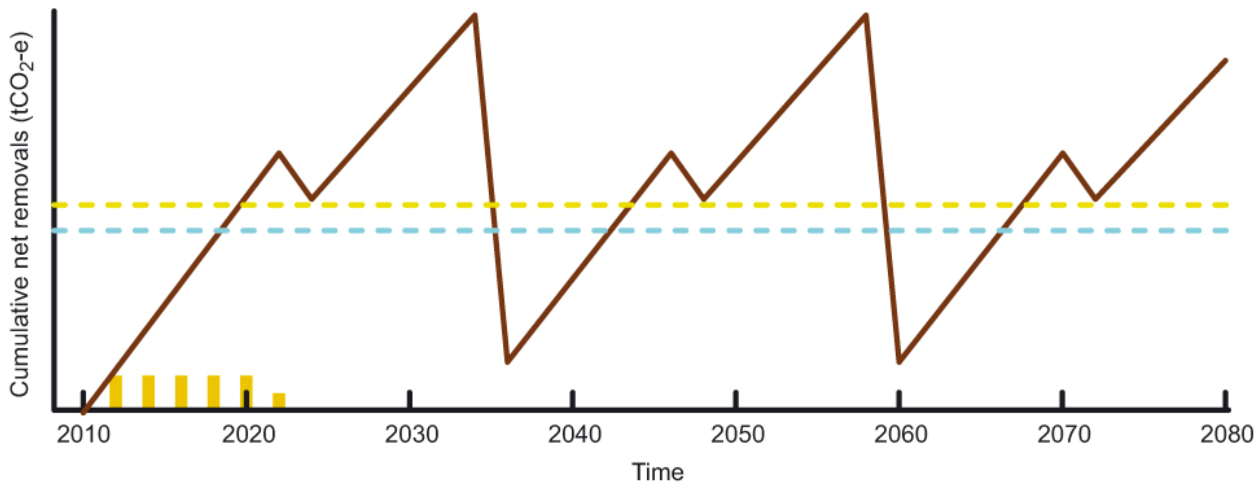

FIGURE 12: EXAMPLE GHG NET REMOVALS FROM A HARVESTED FOREST RE-ESTABLISHED OVER TIME (SOURCED FROM: DEPARTMENT OF CLIMATE CHANGE 2008B)

*The yellow dashed line represents the permit limit and the blue dashed line the 'risk reversal buffer'. The yellow bars indicate the permits issued during the forests growing stages. Note that permits are issued on the long-term basis (for example 70 years). Forest identities will need to surrender permits if the forest is not re-established after harvesting.

6.3.2 Measuring Carbon sequestration

The units used to describe carbon sequestration in a forest sink are CO_2 or CO_2-e. Emissions inventories present emissions for each greenhouse gas as CO2-e enables the comparison of the integrated effect of emissions of the various greenhouse gases, which have different warming effects. Describing carbon sequestration in these units allows for the direct comparison between sequestration and emissions. Multiplying the quantity of carbon by 3.67 converts it into units of CO_2 (based on molecular mass of carbon and oxygen).

Carbon fluxes in a forest system are complex, but net sequestration or emissions can be calculated as changes in carbon stocks over time. Australia uses the stock-change approach for accounting for forest sinks using the formula:

$\Delta C_i = C_i - C_{i-1}$

Where:

ΔC = change in carbon stocks per year

C_i= carbon stocks in a year

C_{i-1} = carbon stocks in the year before year i.

Project-level carbon accounting involves estimating the change in tones of carbon, CO_2 or CO_2-e per ha per year (Department of Environment and Heritage 2006).

Modelling

Project level carbon sequestration for stands of trees can be modelled using the National Carbon Accounting Toolbox (Department of Climate Change 2005) which can estimate net carbon sequestration from reforestation without requiring the input of information on tree volume and wood density. Estimates are based on site characteristics (e.g. soil type and climate), and can be modified to reflect actual management regimes and a set of standardised factors by entering site-specific data such as tree species, year of establishment, thinning events, rotation length, weed management and fertiliser application.

Field Measurements

The data required to model the carbon sequestration in forests is derived from field measurements. The sampling strategy most suitable to estimate the quantity of carbon present in a forest will depend on the forest structure, composition and scale of various stands involved, the objective of the inventory and the resources available for sampling. The carbon pool of a forest system has four different components:

- Carbon in above-ground living biomass
- Carbon in coarse woody debris
- Carbon in surface litter
- Carbon in root systems
- Soil carbon

The biomass of above ground components are generally easier to estimate than below-ground components. A brief overview of the methods commonly used for estimating carbon in each component is below. When reporting carbon sequestration under a carbon trading scheme it will be very important to accurately describe the method used for each forest situation including sampling techniques and any equations or assumptions made in calculations.

Above ground biomass

Overstorey Species

The biomass of larger vegetation is generally estimated by applying a ratio or regression methods to an easily measured variable such as stem diameter. Other variables that are often measured include the diameter at breast height, tree height, stem diameter at base of green crown, crown diameter, bark thickness, bole length and crown length. Several common methods for estimating biomass (adapted from Snowman et al. 2002) include:

- Applying a regression equation specific to individual tree species to stem diameter or other measured variable,

- Applying a generic regression equation to an easily measured variable,
- Estimate from species specific or generic yield tables based on diameter or height measurements,
- Estimate stem volume with standard yield tables, apply a density factor (refer to Ilic et al. 2001) to convert to stem biomass and then apply an expansion factor to estimate whole tree biomass.

Understory species

The estimated biomass of woody understorey shrubs can obtained by directly weighing the biomass of fixed area quadrants or by regression relationships using crown diameter or height. Unfortunately, there are few published relationships for Australian native species (see Appendix 1 in NCAS Protocol for Sampling Tree and Stand Biomass, Snowman et al. 2002). For detailed methodology on estimating above ground biomass, refer to the NCAS Protocol for Sampling Tree and Stand Biomass (Snowman et al. 2002).

Coarse woody debris

Coarse woody debris includes logs and branches on the land surface, stumps and large charcoal pieces that have a cross-sectional diameter of >25 mm. Refer to NCAS methodology (McKenzie et al. 2000) for estimating the carbon pool of coarse woody debris.

Surface litter

Surface litter comprises of **dead leaves**, twigs, branches, insect detritus, animal scats, charcoal, other organic matter and woody debris <25 mm. Refer to the NCAS protocols for carbon estimation in soil litter and coarse woody debris (McKenzie et al. 2000).

Roots

Measuring the biomass of roots is an expensive and time-consuming process that will be constricting in a farm forestry situation. Sometimes it is possible to use a regression equation using stem diameter or another measure to estimate the root biomass, Root-shoot ratios can also be used (Snowman et al 2000).

Converting biomass to carbon content

With the exception of the soil component, applying a factor to estimated quantities of oven dry biomass for each component provides an estimate of the carbon content of the forest system. Snowman et al. (2000) recommends that a lab analysis for carbon concentration in each component of the inventory is undertaken.

6.3.3 Case Studies

To assess the possibility of carbon sequestration on sheep and beef farms as a means of offsetting livestock carbon emissions, four case studies were developed. These were not intended to be highly detailed, but rather to provide an indication of the role vegetation management may play in the livestock industry if livestock emissions were taxed at a later date.

Four case study locations were selected to model the potential area of trees that would be required to offset emissions from cattle or sheep production. The potential amount of above ground biomass that is produced by tree plantations and therefore potential carbon sequestration is largely dependent on rainfall and climatic conditions. The case study sites are located in areas with traditionally high sheep or cattle numbers and are representative of production practices in different rainfall and climatic conditions. The two cattle case studies are in central QLD (Figure 13) and southern NSW (Figure 14) and the two sheep case studies in northern NSW (Figure 15) and southern WA (Figure 16). Refer to Table 11 for the details of each case study site and the assumptions we made for estimating the livestock emissions.

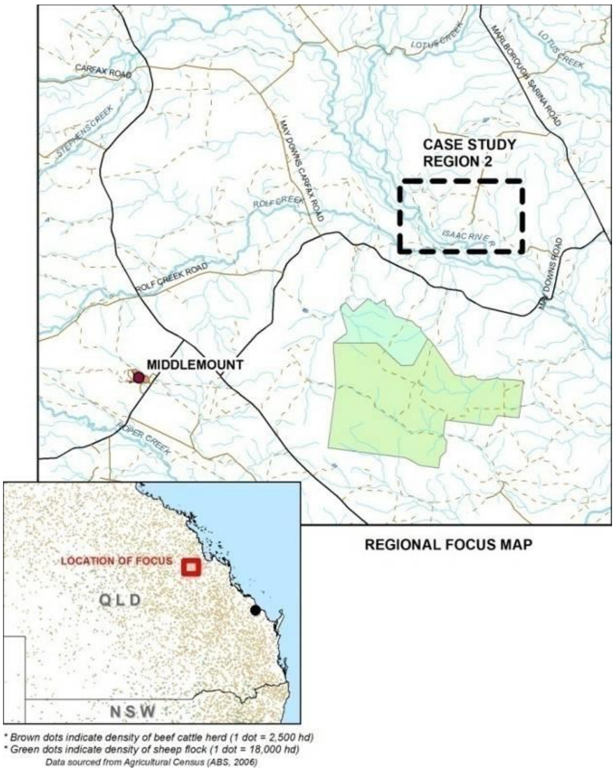

FIGURE 13: CASE STUDY REGION 1 – CENTRAL QUEENSLAND BEEF PRODUCTION

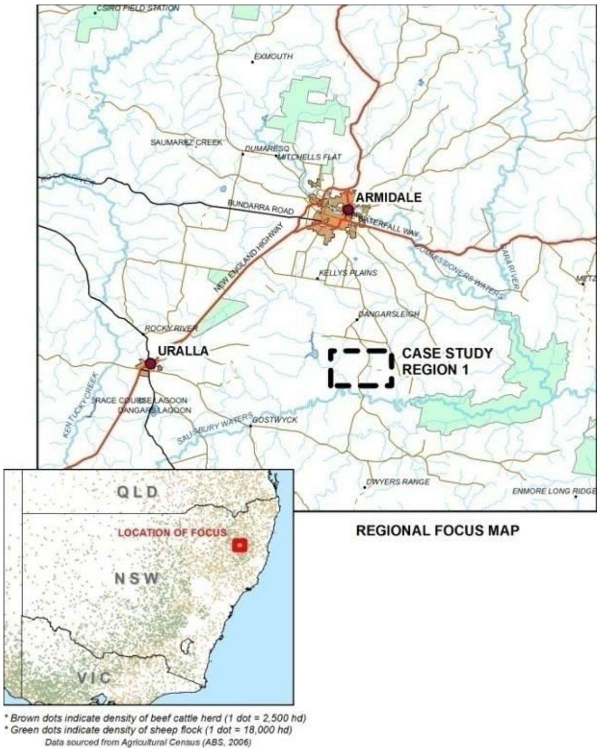

FIGURE 14: CASE STUDY REGION 2 – NORTHERN NSW LAMB PRODUCTION

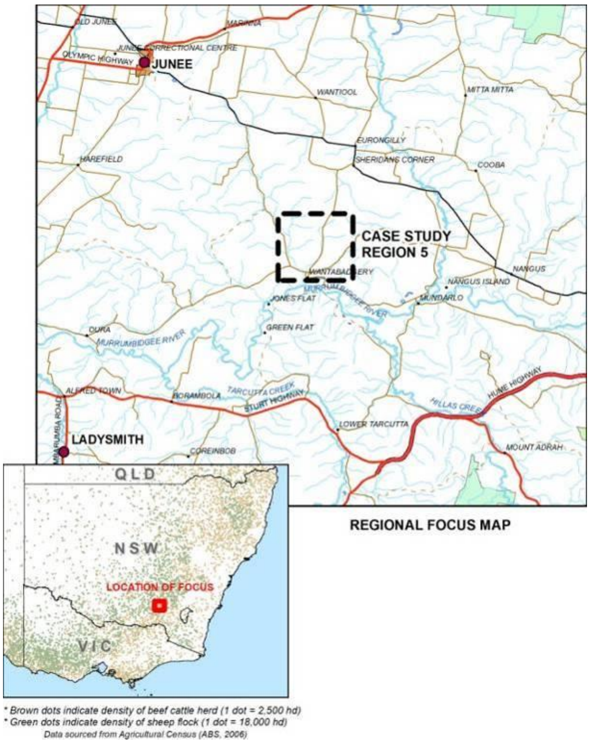

FIGURE 15: CASE STUDY REGION 3 – SOUTHERN NSW BEEF AND LAMB PRODUCTION

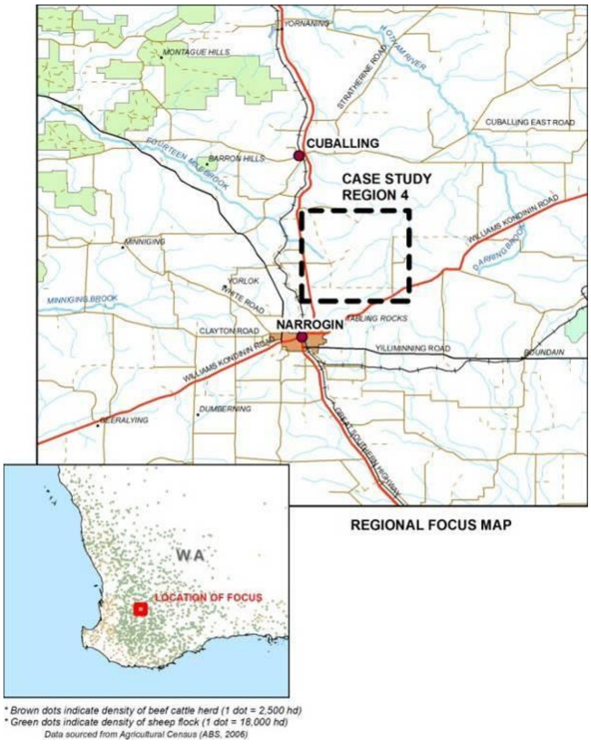

FIGURE 16: CASE STUDY REGION 4 – SOUTHERN WA LAMB PRODUCTION

Methods

Livestock GHG emissions at each case study site were modelled with a farm-scale beef and sheep excel GHG calculator (Eckard at al. 2008). The calculators follow the methodology established by the Department of Climate Change (2007a). The basic assumptions for the case study farms were developed from the general farm practices (i.e. livestock type, stocking rate) for each region studied (Table 11). The only modification to the standard DCC assumptions for livestock emissions was a revision of the livestock growth rates to more closely reflect livestock performance in these regions (based on contact with local producers and expert knowledge).

TABLE 11: LIVESTOCK PROPERTY ASSUMPTIONS FOR THE CASE STUDY FARMS

Assumptions	Case Study Area 1 – Central Queensland (beef)	Case Study Area 2 – Northern NSW (Lamb)	Case Study Area 3 – Southern NSW (beef)	Case Study Area 4 – Southern WA (Lamb)
Land area (ha)	5000	1000	1000	1000
Average rainfall (mm)	650	1000	550	400
Number of breeding animals	1000	4200	440	3000
Number of progeny < 1 yr	900	2800*	400	1800*
Number of progeny > 1yr	440	-	300*	-
Livestock emissions (t CO_2equiv./ year)	5001	1005	1950	625

* Represents the number of head on the farm averaged over the whole year – i.e. if lambs are sold at 8 months age, the average number over 12 months will be the lambs born x 8/12.

Several carbon sequestration scenarios were modelled to offset these emissions. The scenarios used a range of sequestration rates (t CO_2/ha/yr) for different plantings appropriate to the case study regions. The rates of carbon sequestration and the area required to offset emissions is only a guide as the calculator has not been developed to model actual tree growth, local growing conditions or specific management practices. The calculator can model 11 individual tree species and a mixed planting of hardwood or specialty hardwood trees over a thirty-year period. The case-studies rainfall zone (high >700 mm/yr, medium (500-700 mm), low <500 mm) determined which tree species were suitable for each carbon sequestration scenario although due to the low number of options in some cases the species chosen may not be the ideal choice for that location. The rate of carbon sequestration (t CO_2-e/ha/year) in each scenario is a 30-year average sequestration rate for tree growth. Under the proposed CPRS, carbon permits for reforestation would only be issued during the growing-period of the plantation, however trees would have to remain standing the period of the permit (i.e. 70 or 100 years) (Department of Climate Change 2008b). Permits issued for each stand of trees will be based on expected cumulative CO_2 removals for the growing phase of the plantation minus a risk reversal buffer (refer to Figure 11). If for example the risk reversal buffer for a plantation was 10% of the expected cumulative sequestration rate, the property owner would be required to increase the plantation area by 10% trees to offset livestock emissions. This risk-reversal buffer is highly specific and has not been modelled in the case studies.

Consequently, to maintain sequestration rates, new plantings are required at a maximum interval of about 30 years when the trees have reached their peak sequestration potential. To minimise land loss, plantings on a 5 or 10 year rotation may be most effective, though for ease of modelling 30 year planting intervals were assumed for the case study farms, with an overall investigation period of 70 years.

As an alternative method, plantation areas were also calculated using the FarmGAS model (AFI 2009). FarmGAS models the carbon sequestration rate of a ‘mixed environmental planting’ of trees using National Carbon Accounting Toolbox data. A ‘mixed environmental planting’ for a specified region and soil type represents a mixed species, multi-layered forest that is established for the enhancement of farm biodiversity (AFI 2009). It is noted that in some cases a ‘mixed environmental planting’ may not be the best option for sequestering carbon and that other plantation species may be more productive.

Results

The estimated plantation area required to offset current livestock GHG emissions for a 70-year period in each case study is shown Table 12. These represent the 'worst-case' policy scenario where livestock producers are required to offset the total volume of their emissions.

TABLE 12: PLANTATION LAND AREA REQUIREMENTS UNDER FOUR SEQUESTRATION SCENARIOS

Assumptions	**Case Study 1 – Central QLD(beef)**	**Case Study 2 – Northern NSW (Lamb)**	**Case Study 3 – Southern NSW (beef)**	**Case Study 4 – Southern WA (Lamb)**
Land area (ha)	5000	1000	1000	1000
Average rainfall (mm)	650	1000	550	400
Plantation area (ha) (LOW seq. scenario)	1430 (29%)	201 (20%)	782 (78%)	625 (63%)
Plantation area (ha) (MED seq. scenario)	716 (14%)	98 (10%)	663 (66%)	418 (42%)
Plantation area (ha) (HIGH seq. scenario)	446 (9%)	58 (6%)	280 (28%)	250 (25%)
Plantation area (ha) (Mixed Environmental Planting)[1]	3522 (70 %)	310 (31 %)	1043 (104%)	664 (66%)

Note: The percentages in brackets represent the percentage of the total farm area required to offset livestock emissions for 70 years at current livestock emission rates.
[1] Planting area required with mixed environmental planting estimated using FarmGAS (AFI 2009).

The estimated area of tree plantings required were then mapped on a representative area for each location (Figure 17 to Figure 23) with two methods. An exception is the southern NSW site where the FarmGas environmental planting was not modelled because it exceeded the farm area. Two sequestration scenarios are also shown. Areas mapped in orange represent the high sequestration scenarios, while the green represents the additional area required for the lowest sequestration scenario. These maps do not represent particular farms in any way. Please note that under the proposed CPRS (Department of Climate Change 2008b) a larger area may need to be planted (which was not modelled in these examples) to account for the risk reversal buffer.

FIGURE 17: MAP OF CASE STUDY 1 (CENTRAL QLD) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS UNDER A HIGH (ORANGE) AND LOW (GREEN) C SEQUESTRATION SCENARIO.

FIGURE 18: MAP OF CASE STUDY 1 (CENTRAL QLD) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS USING AN ENVIRONMENTAL PLANTING (FARMGAS)

The central QLD (beef) case study shows that tree planting may be a viable option if sufficient sequestration rates are achieved. At the highest sequestration rates plantings along an existing waterway along with two planting strips would be adequate. However, at the lowest sequestration rates (environmental planting, calculated with FarmGas), sequestration with tree planting is not a feasible option to offset emissions. If free permits were provided the viability of sequestration would be greatly improved, depending on the level of permits given.

FIGURE 19: MAP OF CASE STUDY 2 (NTH NSW) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS UNDER A HIGH (ORANGE) AND LOW (GREEN) C SEQUESTRATION SCENARIO.

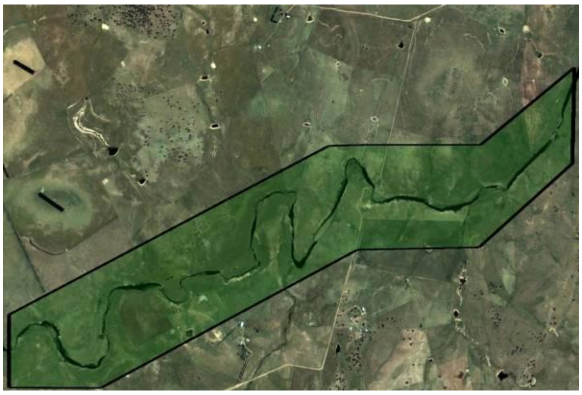

FIGURE 20: MAP OF CASE STUDY 2 (NTH NSW) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS USING AN ENVIRONMENTAL PLANTING (FARMGAS)

The northern NSW (lamb) case study shows a similar pattern to the first case study, where tree planting may be feasible at higher sequestration rates. It is noted that the farming area selected has very little remnant vegetation currently and would benefit from additional trees for shelter. On many farms this will not be the case however, and tree planting would have no additional benefit to the property beyond sequestration. The feasibility of sequestration through tree planting is again very sensitive to sequestration rates. With emission permits, tree planting would be quite attractive in this region of the tablelands, as many farms would benefit from some additional forested areas.

FIGURE 21: MAP OF CASE STUDY 3 (STH NSW) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS UNDER A HIGH (ORANGE) AND LOW (GREEN) C SEQUESTRATION SCENARIO.

The southern NSW case study (beef) showed the lowest feasibility for tree planting to offset emissions, largely because of the relatively low sequestration rates likely in this lower rainfall area and the high level of emissions from a cattle breeding enterprise. Tree planting does not appear to be feasible at this location even with the highest likely sequestration rates. The environmental planting scenario at this site (not shown) exceeded the total farm area.

FIGURE 22: MAP OF CASE STUDY 4 (STH WA) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS UNDER A HIGH (ORANGE) AND LOW (GREEN) C SEQUESTRATION SCENARIO.

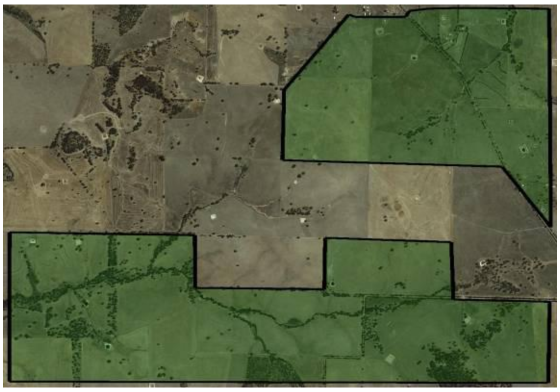

FIGURE 23: MAP OF CASE STUDY 4 (STH WA) SHOWING THE AREA (HA) REQUIRED TO OFFSET CATTLE GHG EMISSIONS FOR 70 YEARS USING AN ENVIRONMENTAL PLANTING (FARMGAS)

The southern WA case study (lamb) shows high land requirements to offset emissions with tree planting. In this region sheep are typically run in mixed farming enterprises (with wheat) making tree planting less attractive because of the immediate loss of crop growing potential.

6.3.4 Mitigation of Carbon with Vegetation on Red Meat Properties

The livestock emissions from cattle in case study 1 (central QLD) and 3 (southern NSW) are equivalent to 1 and 1.9 t CO_2e/ha/yr. Case study 1 has a stocking which is nearly 4 times lower than case study 3 which partly explains the lower rate of livestock emissions of the central Queensland site. To offset current livestock emissions rates for 70 years, the area of land required for tree planting will range from 9% to 70% for the central QLD property and from 28% - 104% for the southern NSW property.

The emissions from sheep in case study 2 (northern NSW) and case study 4 (southern WA) are equivalent to 1.95 and 0.63 t CO_2e/ha/yr respectively. The higher emissions per hectare in northern NSW can be attributed to the higher stocking density. Plantings on these farms ranged from 6% - 31% for the northern NSW farm and 25% - 66% for the southern WA farm depending on sequestration rates.

Planting a large proportion of land to trees could potentially have a large impact on the properties production capacity and income. The potential to offset GHG emissions using trees is largely dependent on the rainfall and the type of species that can be grown. Species with higher rates of sequestration can generally only be grown in high rainfall regions. Farms with high stocking rates in lower rainfall regions will be highly disadvantaged compared to regions with higher rainfalls. Furthermore climate change predictions indicate that most farmland will receive less rainfall and less consistent rainfall events in the future which are required to grow trees with a large biomass potential. If this is the case farming livestock in the lower rainfall regions is a less viable option

6.4 Conclusions, Knowledge Gaps and Recommendations

6.4.1 Conclusions

Vegetation management is one of the most significant issues facing the red meat industries as a result of carbon sequestration opportunities and risk of legislation that further restricts the rights of the land holder to vegetation management and carbon sequestration relating to vegetation growing on private land.

Land use and vegetation management influence the red meat industries from two directions, i) through the effect of legislation on grazing lands which may lead to lower grazing productivity, reduced land value and an increased management and regulatory burden for farmers, and ii) through the opportunities that land managers have for offsetting livestock emissions through sequestration in carbon.

The extent of the risk can be seen through the Queensland Government's moratorium on all vegetation clearing, which is currently in place. If this policy is enshrined in legislation, graziers can expect to face decreasing stocking rates and land values as vegetation re-grows, while not having the right to claim the value of carbon sequestered on their own land.

Despite tightened controls on vegetation management, the red meat industries may have the capacity to sequester a large amount of atmospheric carbon to offset emissions where land has been cleared prior to 1990. Moreover, the industry will play a key role in contributing to emissions abatement at a national level, and can expect recognition for this role. For these reasons it is in the long-term interests of the red meat industry that it investigates and implements effective practices to reduce GHG emissions and enhance the amount of carbon it sequesters. It is estimated that about half of Australia's annual emissions could be absorbed by improved land management practices that enabled revegetation in Australia's arid and semi-arid rangelands. Upgrading of savanna management also has substantial mitigation potential, and would also have positive effects for biodiversity conservation. This will have implications on the northern Australian Beef Industry, a significant component of Australia's red meat industry where land areas a still relatively undeveloped. Producing biomass as a feedstock for biofuels may also be a possibility in the future.

A number of case studies were developed to give a basic visual representation of the land area required to offset GHG emissions from livestock. These case studies showed variable results, which may be practical in some farming situations (such as the northern NSW sheep property) provided adequate sequestration rates could be achieved. If livestock managers were given substantial proportions of free permits (up to 90%), sequestration may allow the remaining emissions to be effectively offset, though this could still result in up to 10% of the property area being required over a 70 year period. The feasibility of tree plantations will also depend on the associated benefits that may be provided to the landholder through provision of ecosystem services (particularly in dryland salinity areas) or possibly diversified income through agro-forestry.

6.4.2 Knowledge Gaps and Recommendations

Key knowledge gaps that exist in this area are:

- The real and potential impacts of vegetation management laws on the viability of grazing enterprises (reduced carrying capacity, reduced land value – particularly in Queensland where a recent moratorium on all vegetation management has been imposed)
- The value of reduced emissions / sequestration from the grazing sector to the Australian community both on an industry basis and on a 'per kilogram of product' basis to overcome the division of land use and livestock emission estimation
- The potential rate of sequestration (tonnes CO_2-eq / ha / year) for a range of livestock production regions in Australia
- Regulatory frameworks to ensure fair treatment of on-farm sequestration
- Cost effective measurement of new and in-situ plantations
- The opportunities for sequestration in agro-forestry ventures (where product is harvested for profit)
- The costs of compliance involved with on-farm sequestration through vegetation management

Recommendations for further research in this area are as follows:

- Detailed evaluation of the carbon sequestration potential via vegetation and soils for a range of production regions, incorporating the economic impact of a range of sequestration options.
- Quantification of the current GHG sequestration occurring on land used for red meat production as a result of changes to vegetation management.
- Research into dual purpose woody pastures such as luceana to quantify the sequestration and production potential.

7 Review of GHG Emissions from Different Protein Sources

7.1 Introduction

Red meat is frequently criticised for its greenhouse gas performance by organisations and researchers that promote alternative products, primarily of plant based proteins. Notwithstanding the methodological differences between studies, it may be possible to arrive at a method for comparing dissimilar products using a robust environmental impact tool such as LCA. However, this raises a variety of issues to manage, as each protein source has a different protein composition and digestibility, and contains different amounts of energy, minerals and vitamins. These differences are especially apparent when comparing plant and animal protein sources. Despite this, a small number of studies have attempted to compare the environmental impact of different dietary protein sources. For example, Davis et al. (2008) compared the energy requirements and GHG emissions of four meals containing different proportions of grain legumes but equal proportions of protein and energy.

7.2 Beef

The environmental impact of producing beef for human consumption has been investigated by various lifecycle assessments, material flows, carbon footprint and food mile studies. Most studies indicate that the production phase of red meat, and in particular beef, contributes a large percentage to the environmental impact of the end product (i.e. Peters et al. 2009a). This review will compare the greenhouse gas emissions produced by the production stage (cradle to processing) of Australian beef, to beef produced in the EU and seven other countries. The respective studies cover different production systems including feedlot and pastoral production, organic and conventional farming practices, and extensive versus intensive systems.

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 report energy requirements (Barber and Lucock 2006) for beef production. Barber et al. (2007) investigated tallow from beef production. Some studies investigate the GHG emissions of beef as part of a meal (i.e. Sonnesson et al. 2005a).

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 IPPC 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) is 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 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) and Barber et al., cited in Goldberg (2008) 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 LW 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 study 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 where pasture based and did not include grain feeding. These studies report a wide range in GHG emissions from 8.4 kg CO_2e/kg CW (Sahelian), 18.1 kg CO_2e/kg CW (AUS), 22.2 CO_2e/kg CW (EU) and 28 kg CO_2e/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 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) this appropriateness of this approach is questionable. To date, few studies have incorporated alternative scenarios based on ‘best practice’ management for the reduction of major greenhouse gases 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 13). 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 of 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 14). 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 14). 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 (18.1 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 (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 14). 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 LW in 1981 to 10.4 kg CO_2e/kg LW 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 13: 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		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	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**

[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 14: 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)	organic (2004)	20.2
Peters et al. (2009a)	AUS (NSW)	grass/feedlot (2002)	24.4
Peters et al. (2009a)	AUS (NSW)	grass/feedlot (2004)	20.0
Barber et al. (2007)	NZ		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**
Weidema et al (2008b)	EU-27		276

7.3 Lamb and Sheep Meat

There are fewer studies that review the GHG emissions from lamb or sheep meat production. This review compares the GHG emissions from sheep meat production (cradle to processor) from four countries including Australia. Other studies have calculated the energy requirement for the production of sheep meat (Barber and Lucock 2006; Schlich and Fleissner 2005). Saunders et al. (2009) report the CO_2 emissions and the energy requirements of New Zealand and UK sheep meat production. Barber and Pellow (2006) have also studied the total energy requirement and CO_2 emissions of the New Zealand wool production systems.

LCA Methodology and System Boundaries

The majority of the studies follow an LCA methodology based on ISO and IPPC standards. Nemry et al. (2001) used a materials flow approach to calculate GHG emissions using the CORELLI model. The system boundaries for each study differ. The Australian system boundary incorporates emissions from cradle to the processing phase (Peters et al. 2009), the Belgium study (Nemry et al. 2001) is from cradle to retailer and the UK/Wales study (Williams et al. 2006) is from the cradle to the farm-gate (does not include meat processing). The New Zealand study by Goldberg (2008) expands the results of Barber et al. (2007) to include transport to the meat processer and processing, and transport to a London port (the figure included in Table 15 only includes transport to processing and the processing emissions). As with the beef systems, the majority of emissions are associated with the production and not the processing stage of the system.

Management of co-products

As with the beef studies, a number of methods are applied to manage co-products (i.e. point of slaughter and wool). As with beef, the method of handling co-products has a significant effect on the results of the study and can make comparison between studies difficult.

Comparison between countries

The quality of data and the extent of inventory of each study vary so comparing the results of each study is difficult. For example, three studies included the embodied energy and emissions from the production of farm machinery and/or buildings (Barber et al 2007, Williams et al, 2006, and Peters et al. 2009) and Nemry et al. (2001) includes emissions for breeding sheep (which are quite substantial) but not specifically for animal production (enteric methane production and manure management). The main GHG contributor in the sheep production system is methane (CH_4), followed nitrous oxide emissions (N_2O) in the Australian and New Zealand case studies. The UK/Wales study does not break down its GHG emissions into separate components to allow for comparison. The Belgium case study (Nemry et al. 2001) did not conduct a full LCA, only a material flow analysis, so it appears that enteric methane production, manure management and N_2O emissions were not calculated for the production phase, which may explain why the biggest contributor is CO_2.

Production systems

Only one study directly compared organic and non-organic production systems (Williams et al. 2006). The organic production of sheep meat had lower overall GHG emissions compared the non-organic system. When the economic allocation was altered to give mutton a higher value (£30 to £100 per animal) the GHG of the non-organic system was reduced by nearly 17%. Interestingly the organic system required more land, 3.12 ha/t of sheep meat versus 1.38 ha required to produce one tonne of non-organic sheep meat. The organic system also had higher eutrophication and acidification impacts. The NZ production systems are not detailed in the literature. The Australian system in based on a low rainfall pasture system with supplementary grain feeding to finish lambs prior to slaughter.

TABLE 15: GLOBAL WARMING POTENTIAL OF LAMB AND SHEEP MEAT PRODUCED FROM DIFFERENT COUNTRIES AND PRODUCTION SYSTEMS ASSESSED USING LCA

Reference	Country	system	GHG	kg CO_2e/kg CW unallocated[1]	system boundaries
Goldberg (2008)	New Zealand		**total**	**19.5**	cradle to processing
Nemry et al. (2001)	Belgium		CH_4	1.9	cradle to retail
			N_2O	7.6	
			CO_2	9.3	
			total	**18.8**	
Peters et al. (2009a)	AUS (WA 2002)	Pasture/some supp feeding	**total**	**10.8**	cradle to processing
	AUS (WA 2004)	Pasture/some supp feeding	**total**	**10.2**	cradle to processing
Williams et al. (2006)	UK/Wales	non-organic	**total**	**20.1**	cradle to farm-gate
		organic	**total**	**11.6**	cradle to farm-gate

[1] For comparison between studies, data have been re-analysed to attribute no allocation of environmental burden to useful by-products at the point of slaughter. In several studies the allocation processes used were not clear, but wherever possible results were checked by re-analysing primary data. The reader is directed to the original references for further information.

7.4 Alternative Protein Sources

7.4.1 Pork

Intensive pig production is often associated with adverse environmental impacts related to production at the farm level. Several LCA studies have been done for various systems of pork production, primarily in Europe. These studies have assessed environmental impacts of global warming, eutrophication, and acidification (Basset-Mens & van der Werf 2005 and Dalgaard 2007, Dalgaard et al. 2007a) as well as energy use, land use and pesticide use (Basset-Mens & van der Werf 2005; Cederberg & Flysjo 2004a). None of these studies assessed water use.

All three studies focused on identifying the environmental 'hot spots' in the pork production chain, although the functional units and the goals for each study were usually quite different.

Dalgaard et al. (2007a) assessed the environmental impacts for 1kg of Danish pork (carcass weight) delivered to the Port of Harwich in England. The goal of the report was to compare the environmental impacts of 1 kg of Danish pork delivered to Port Harwich to an equivalent kg of pork produced in Sweden, France and Great Britain.

The highest contributions to global warming for their study arose from the production of feed and the handling of manure. Interestingly, the greenhouse gas contribution of transport (i.e. the 'food miles') was minimal largely because the mode of transport was by ship, which has a low level of CO_2 intensity per tonne km (tkm) travelled. A limitation with the concept of 'Food miles' is that the mode of transport is often overlooked and there are considerable differences in CO_2 emissions between transport by ship or by truck.

Dalgaard et al. (2007a) noted that GHG emissions could be reduced for pork production through lowering feed (and protein) consumption and improving the handling of manure/slurry. In particular, greenhouse gas emissions can be reduced if the manure/slurry is treated in an anaerobic digester, with the biogas used for heat and power production.

The objectives of the Basset-Mens & van der Werf (2005) study were to evaluate the environmental impacts of three contrasting pig production systems. The scenarios compared were conventional good agricultural practice (GAP) according to French production rules, a French quality label scenario called red label (RL) and a French organic scenario called organic agriculture (OA). In the GAP production system pigs are raised at high density in a conventional slatted-floor building similar to pig production in Australia.

Energy data were also presented by these authors, showing the fairly heavy reliance on energy for production (Table 16). Energy use is consistently higher than for beef production in Australia. This trend may not be as apparent if Australian pork energy usage data were available however, because a large proportion of energy usage is derived from grain production in Europe, which is far more intensive than Australian grain production.

TABLE 16: COMPARISON OF GLOBAL WARMING POTENTIAL AND PRIMARY ENERGY USE FROM THE PRODUCTION OF ONE KILOGRAM OF PORK ASSESSED USING LCA

Reference	kg CO_2-e / kg Carcass weight[1]	MJ Primary Energy / kg Carcass weight
Dalgaard et al. (2007a)	**3.3**	**-**
Basset-Mens & van der Werf (2005)	**3.02**	**15.9**
Cederberg & Flysjo (2004a)	**4.4**	**18.4**
Williams et al. (2006)	**6.4 (non organic)** **5.6 (organic)**	**15.8 (non organic)** **18.2 (organic)**
Weidema et al. (2008b)	**11.2 (Slaughter Weight)**	**193**

[1] where necessary, data were transformed to a carcass weight basis (unallocated) for comparison purposes.

It is noted that at the time of writing, an Australian pork LCA project has been completed which will present results that are more comparable to Australian red meat. It is recommended that the reader source this document when it is released publically for review by Australian Pork Limited.

Pork production is generally lower in GHG emissions because of the lower livestock emissions related to monogastric digestion compared to ruminants. Consequently, the primary driver of emissions is methane and nitrous oxide from the waste management system (effluent and solid manure) and from the production of grain crops fed to the pigs. It is important to note that pigs rely wholly on grain inputs for production. Hence, arable land use will be a significant factor for comparison with extensive red meat production. However, this is outside the scope of the current study.

7.4.2 Chicken Meat

Relatively few studies have investigated the environmental impact of poultry meat production and in particular the emissions of GHG or water use. A review of the literature discovered five studies that had estimated the GHG emissions from the production phase of the meat chicken supply chain (Table 17). Of these, three are comprehensive LCA studies.

Two studies calculated the GHG emissions associated with a meal that contained chicken. Davis & Sonesson (2008b) estimated that 1.7 kg of CO_2-e emissions were related to the production phase of a kilogram of chicken meat used in a homemade meal, while Katajajuuri (2007) attributed 3.1 kg CO_2e to the production and processing stage of a chicken fillet for retail. In other studies, Elferink & Nonhebel (2007) calculated that 7.7 m^2 of land was required to produce 1 kg of poultry carcass weight in the Netherlands. For the purpose of this review, only studies that investigated GHG emissions were reviewed in detail. No studies including water use were found.

TABLE 17: COMPARISON OF GHG EMISSIONS PRODUCED DURING THE PRODUCTION OF MEAT CHICKENS ASSESSED USING LCA

Reference	Country	System description	GHG	kg CO_2e/kg CW[1]	System boundary
Bennett et al. (2006)	Argentina	Non-organic	**total**	**3.1**	cradle to farm-gate
Nemry et al. (2001)	Belgium	-	CO_2	0.8	cradle to retailer
		-	CH_4	0.7	
		-	N_2O	0.7	
		-	**total**	**2.1**	
Pelletier (2008)	USA	Non-organic	**total**	**2.0**	cradle to farm-gate
Williams et al. (2006)	UK/Wales	Non-organic	**total**	**4.6**	cradle to farm-gate
		Organic	**total**	**6.7**	
		Free-range	**total**	**5.5**	
Weidema et al. (2008b)	EU-27	-	**total**	**3.6**	cradle to grave

[1] where necessary, data were transformed to a carcass weight basis (unallocated) for comparison purposes.

All of the studies used LCA methodology to determine GHG emissions, with the exception of Nemry et al. (2001) who analysed material flows to calculate the indirect GHG emissions from poultry. This may explain why the emissions from this study are lower than those calculated for the other studies. The system boundaries in all studies were from cradle to the farm-gate, with the exceptions of Weidema et al (2008b) whose system boundary extended to the grave, and Nemry et al. (2001) which extended to the retailer.

The production systems investigated varied greatly across studies. Williams et al. (2006) compared organic, free range and non-organic systems, while Weidema et al. (2008b) studied all production systems present in a global region (the EU-27) which were then used to create a weighted average for all chicken production in the region. Other studies (Nemry et al. 2001) did not specify what production system is used. Results from Williams et al. (2006) suggested that organic production systems produce more GHG emissions and have a higher energy and land requirement than non-organic (barn) system or free-range systems. These results were attributed to the organic chickens having a poorer feed conversion ratio and longer growing periods.

Each study employed slightly different life cycle assessment methods and used a different inventory methodology, making comparisons difficult. However, some trends seem apparent. The production of feed is the biggest contributor to GHG emissions for poultry. In the US study, feed production was responsible for 82% of GHG emissions, 98% of ozone depleting emissions and 80% of supply chain energy use (Pelletier 2008). This has led some researchers (i.e. Bennett et al. 2006) to compare specific grain production scenarios to determine the effect this has on the overall performance of meat chicken production. Bennett et al. (2006) discovered that the use of GM corn produced 13.5% less GHG emissions compared to non-GM corn, which could be significant if diets were based primarily on corn.

Meat chicken is a low GHG burden meat, though not as low as may have been expected considering the superior feed conversion efficiency and low direct animal emissions (i.e. enteric emissions) from chickens. It is likely that GHG from energy use (particularly for heating and cooling) are a contributor to this, and that the poor GHG performance of grain production in the study regions will skew the results compared to production under Australian conditions. However, this has not been tested to date by an Australian LCA study.

7.4.3 Dairy Products

Dairy products and supply chains have been extensively studied using LCA, with more studies presented for various dairy products than any other food group. Despite this, differences in methodology and the assumptions made between studies make it difficult to directly compare the GHG emissions of different dairy products (Basset-Mens 2008).

Comparison of dairy products as an alternative source of protein (i.e. on a nutritional basis) to red meat depends on the protein content of different dairy products. The protein content of some common Australian dairy products is summarised in Table 18.

TABLE 18: PROTEIN CONTENT OF COMMON AUSTRALIAN DAIRY PRODUCTS

Dairy Product	**Protein (%)**
Milk	
Regular Milk	3.3
Skim Milk	3.6
Low Fat	4.6
Yoghurt	
Regular Yoghurt	4.7
Low Fat Yoghurt	5.9
Cheese	
Cheddar	25.3
Mozzarella	26.9
Brie	19.3
Ricotta	10.5
Cream Cheese	7.6

Source: Australian Dairy Corporation (1999).

Cheese has the highest protein content of all dairy products. In comparison, raw red meat contains between 20-25% protein, of which 94% is digestible, while cooked red meat has between 28-36% protein (Williams 2007).

Milk

A recent methodological review of the carbon footprint of raw milk from the cradle to the farm gate has been conducted by Bassett-Mens (2008). The results from this review for GHG emissions for the production of 1 kg energy corrected raw milk (ECM) at the farm gate are presented in Table 19. Eide (2002), Berlin (2002) and Hospido et al. (2003) have also conducted studies on raw milk production which extend the system boundary beyond the farm gate, and were therefore not included in the Basset-Mens (2008) review. These studies found that the production stage (on-farm) of the milk supply chain was responsible for the majority of GHG emissions for either milk (Eide 2002, Hospido et al. 2003) or cheese (Berlin 2002) production through to retail.

TABLE 19: COMPARISON OF GHG EMISSIONS (GLOBAL WARMING POTENTIAL) PRODUCED DURING THE PRODUCTION OF MILK ASSESSED USING LCA

Reference	Country	Allocation rules	System Description	GWP (kg CO_2e/kg ECM*)
Basset-Mens et al. (2008)	New Zealand	Based on fodder requirement (i.e. 85% of burdens allocated to milk, 15% to meat);	Average New Zealand System	**0.86**
Casey and Holden (2005)	Ireland	Economic allocation between milk and meat	Average Irish System	1.3
Cederberg & Mattsson (2000)	Sweden	Fodder requirement (85% milk, 15%meat); mass allocation land area; economic allocation feed ingredients	Conventional	0.95
			Organic	1.10
Cederberg & Flysjö (2004b)	Sweden	Economic allocation (90% milk, 10% meat)	Conventional high	0.9
			Conventional medium	1.04
			Organic	0.94
Haas et al. (2001)	Germany	None (but meat production est. to be 10%)	Conventional Intensive	1.3
			Conventional Extensive	1
			Organic	1.3
Thomassen et al (2008)	Netherlands	Economic (conventional – 91% milk, 8.2% animals, 0.8% exported crops; organic – 90% milk, 6.6% animals, 3.4% exported crops and manure	Conventional	1.41
			Organic	1.48
Williams et al (2006)	UK/Wales	Economic for milk and feed ingredients; maintenance cost of cows avoided when dairy bred cows enter beef sector; 50% avail N in slurry used to save fertiliser	Weighted average between conventional, organic and alternative production systems	1.03**

*: ECM = Energy Corrected Milk – also called fat and protein corrected milk (FPCM)
**: GWP result expressed per kg fat-corrected milk from a result of 10.6 per $10m^3$ fat-corrected milk

All studies completed a comprehensive LCA cradle to farm-gate analysis of raw milk production under typical conditions of production at the national level for the country studied. Several studies also explore alternative production and management options such as conventional and organic production. The functional unit for the majority of studies was 1000kg or 1kg of Energy Corrected milk (ECM), with the exemption of Haas et al. (2001) who didn't specify milk quality and Williams et al. (2006) who used a far larger volume as a functional unit ($10m^3$) of fat-corrected milk.

The origin of data used for the LCA to represent a "typical scenario" of a dairy farm in a specific country was either based on a small number of real farms (Cederberg and Mattsson 2002, Cederberg and Flysjö 2004b, Haas et al. 2001, Thomassen et al 2008) or from national statistics (Casey and Holden 2005, Williams et al. 2006, Basset Mens et al. 2008). Both methods have advantages and disadvantages. Using a sample of real farms may skew results if these farms are not representative, particularly where data are collected from "industry leaders" who employ best management practices. However, national data sets are often incomplete and LCA studies based on national data require other sources of information to complete an inventory, potentially introducing inconsistencies within the data.

Environmental burdens associated with milking cows that enter the beef industry were allocated by either biological causality (fodder requirement) Cederberg and Mattsson (2000) and Bassett-Mens et al. (2008), or economic allocation (Cederberg and Flysjö 2004b, Casey and Holden (2005), Thomassen et al (2008) and Williams et al 2006). Haas et al. (2001) used no specific rules but recognised meat production as a significant co-product and estimated it to amount to some 10% of the burden. In the studies that specified the allocation weightings, 85% milk and 15% meat was typically used for biological causality, and 90% milk / 10% meat was used for economic allocations.

The inventory methods used to estimate GHG emissions from methane and nitrous oxide also varied between the seven studies. Generally, local references were favoured over more general references (such as the IPCC); however in most studies a combination of methods was employed. The exact method applied was not always clear from the publication however.

Most studies modelled an average or conventional dairy system, often compared to organic production systems and/or to systems with differing levels of production intensity. The results (Table 19) indicate that organic production systems generally produce more GHG emissions than conventional systems and more intensive dairy operations produce similar (Cederberg and Flysjö 2004b, Haas et al. 2001) or higher GHG emissions (Basset-Mens et al. 2008) than less intensive systems.

The results between the 6 countries are fairly consistent (0.85-1.4 kg of CO2e/kg ECM). However, despite the apparent consistency Bassett-Mens et al. (2008) did not believe this was sufficient evidence to confirm the relative performance of milk production from one study to the next or in general, considering the lack of consistency and transparency between the assumptions and methods employed in each study.

Cheese and Yoghurt

Fewer LCA studies have been conducted on the production of processed dairy products such as cheese and yoghurt. Berlin et al. (2002) conducted a study on the production of a semi-hard cheese in Sweden using the milk production LCA results from Cederberg and Mattsson (2002). The production of 1 kg of Ängsgården (semi-hard cheese) wrapped in plastic produced 8.8 kg CO_2e GHG emissions, of which 94% was attributed to milk production. Enteric methane was the greatest single source of GHG emissions. The report noted that, as Sweden uses predominantly non-fossil fuel sources for electricity generation, CO_2 emissions from electricity use were much lower than the European average. This would have the combined effect of reducing overall GHG emissions and increasing the apparent influence of livestock emissions. Considering the lack of published research, it is difficult to make accurate comparisons of processed dairy products to meat. However considering the yield of cheese from fresh milk is around 10% by mass (Nielsen 2006), cheese is likely to compare favourably with red meat per unit of protein.

7.4.4 Comparative Studies – Meat Products

Comparison between meat products must take into account the stage of supply and the physical and quality factors that influence the product. For example, beef and pork are not comparable as carcass weight, as pigs are commonly handled 'head on, skin on' and therefore have a considerably higher dressing percentage than beef. At the retail level however, a consumer may naturally choose between buying a chicken breast, beef steak, lamb cutlet or pork chop for the evening meal, implying that at the retail / consumer level, these products are roughly comparable. This being said, the products must again be balanced for quality. This is somewhat subjective however. Table 20 attempts to compare the GHG emissions from beef, sheep, pork and chicken as carcass weight averages from the reviewed literature in previous sections, even though as previously mentioned this may not be quite correct. However the results do indicate a general trend in that beef production produces the most GHG emissions, followed by sheep meat, pork and lastly chicken. There is a large variation in GHG emissions within individual meat products, which suggests that there is scope to reduce emissions from all sources. For example, the lowest beef emissions recorded in the literature are lower than the highest pork emissions recorded.

Another option is to take a meal as the point of comparison. This may introduce differences in the portion size of alternative products as served. When a comparable product has been determined, additional complications may still be introduced because of the varied methods adopted for LCA modelling of different products, making comparison meaningless. The most useful studies for the comparison of meat products are those done within the same project, using the same methodology and data collection practices. Examples of such studies include Weidema et al. (2008b) and Williams et al. (2006). Weidema et al. (2008b) compared 4 animal based food groups commonly consumed in Europe. This study showed that on a per kg basis, beef had 4-8 times the environmental impact of poultry meat production and 5 times the impact of pork production.

Baumgartner et al. (2008) presented an interesting paper comparing the environmental impacts of different diets for animal production, including beef, pork and poultry meat. This study showed that by manipulating feed ingredients (substituting products known to generate high emissions because of production system or distance of transportation); significant (10 %) reductions in global warming potential of the meat product could be made.

TABLE 20: COMPARISON OF A RANGE OF MEAT PRODUCTS BASED ON GLOBAL WARMING POTENTIAL PRESENTED IN THE LITERATURE

	Global Warming Potential		
	Average (kg CO_2-e kg Meat[1])	**Range (kg CO_2-e kg Meat[1])**	
Beef	18.7	8.4	28.7
Lamb	15.2	10.2	20.1
Pork	5.9	3.0	11.2
Chicken	4.2	2.0	6.7

[1] Results presented on an 'unallocated basis' meaning all burden has been transferred to the meat carcass.

7.4.5 Plant Protein Sources

The diet of Western countries is heavily dependent on meat and dairy products, and many groups suggest that substituting a higher proportion of the diet with plant protein would significantly reduce GHG emissions, water use and other environmental impacts associated with food production. This has been supported by relatively fewer scientific studies, though some do exist (i.e. Reijnders & Soret 2003; Carlsson-Kanyama et al. 2003).

Comparing the environmental impacts from meat to equally nutritionally balanced plant-protein alternatives is challenging as plant sources contain different proportions of digestible amino acids to meat. Consequently, a plant-based protein alternative to meat must contain a mixture of complementary plant protein sources to achieve the right balance of amino acids. Soybean protein is the most complete plant protein source, although it is deficient in the essential amino acid methionine. This review covers LCA studies of alternative plant products, and of 'balanced meals or diets'.

Plant protein sources

Very few studies have evaluated the environmental impact of plant protein sources which include a wide variety of foods, for example soy products such as tofu, tempera and textured soy protein, beans, lentils, barley, brown rice, buckwheat, millet, oatmeal, quinoa, rye, wheat germ, wild rice, large range of vegetables and fruit and various nuts and seeds. Two studies were found that evaluated the life cycle assessment of soy beans to the farm-gate (Carlsson-Kanyama et al. 2003, Williams et al. 2006) but none were found for tofu or other processed soy products. Soya beans produced in Argentina were associated with 50% less GHG emissions than soya beans produced in the UK/Wales (Williams et al. 2006). This could be associated with higher fertiliser application rates in the UK, as the Argentinean study assumed that no nitrogen was applied. Williams et al. (2006) have evaluated the LCA of the production of several grain crops and potatoes to the farm-gate, and a Netherlands study has determined the GHG emissions associated with a large list of food products available for retailer (Kramer et al. 1999).

Davis and Sonesson (2008a) compared a 'pea burger patty' with a pork sausage, ensuring that both meals had an identical protein and carbohydrate content. The results showed that the vegetarian meal resulted in 40-80% lower impacts for most impact categories (eutrophication, acidification, greenhouse gas); however the results were roughly equal in terms of primary energy use because of the extensive processing required for production of the pea burger. Because of this, the overall GHG benefit of the pea burger compared to the pork sausage meal was in the order of 30%. It is possible that in Australia, where greater GHG emissions are generated during primary energy (electricity) production compared to Europe, that this difference would be lower still. It was also noted by the author that the meals do not provide the same nutrition, particularly in relation to amino-acid profile.

Some of these examples are presented in Table 21. While not comparable to beef, these data are still interesting when considering the environmental profile of a balanced diet.

TABLE 21: EXAMPLES OF GHG EMISSIONS FROM PLANT FOOD PRODUCTS ASSESSED USING LCA

Plant-based protein sources	Country	kg CO_2e/kg product	System Boundary	Reference
Soya beans	Argentina	0.64	cradle to farm-gate	Carlsson-Kanyama et al (2003)
Soya beans	UK/Wales	1.3	cradle to farm-gate	Williams et al (2006)
Dry peas	Sweden	0.68	cradle to retailer	Carlsson-Kanyama (1998)
Barley	UK/Wales	0.72	cradle to farm-gate	Williams et al (2006)
Bread wheat	UK/Wales	0.80	cradle to farm-gate	Williams et al (2006)
Rice	The Netherlands	36.51	cradle to retailer	Kramer et al. (1999)
Rice	Sweden (imported)	6.4	cradle to retailer	Carlsson-Kanyama (1998)
Potato flour	The Netherlands	1.71	cradle to retailer	Kramer et al. (1999)
Wheat meal	The Netherlands	8.99	cradle to retailer	Kramer et al. (1999)
Potatoes	The Netherlands	49.24	cradle to retailer	Kramer et al. (1999)
Potatoes (non-organic)	UK/Wales	0.22	cradle to farm-gate	Williams et al (2006)
Potatoes (organic)	UK/Wales	.20	cradle to farm-gate	Williams et al (2006)
Potatoes	Sweden	0.17	cradle to retailer	Carlsson-Kanyama (1998)
Spinach	The Netherlands	2.1	cradle to retailer	Kramer et al. (1999)
Fresh pulses/leguminous plants	The Netherlands	1.3	cradle to retailer	Kramer et al. (1999)
Vegetables canned or bottled	The Netherlands	48.92	cradle to retailer	Kramer et al. (1999)
Carrots	Sweden	0.5	cradle to retailer	Carlsson-Kanyama (1998)
Carrots and tubers	The Netherlands	1.81	cradle to retailer	Kramer et al (1999)
Tomatoes (glasshouse)	Sweden	3.3	Cradle to retailer	Carlsson-Kanyama (1998)
Nuts and peanuts	The Netherlands	39.49	cradle to retailer	Kramer et al. (1999)
Peanut butter	The Netherlands	9.57	cradle to retailer	Kramer et al. (1999)

It is interesting to note from this table that while some horticulture and crop products have low emissions as expected, others have considerably higher emissions than meat. It must be noted however that the main study represented here (Kramer et al. 1999) relied on broad scale, sector economic data. This was done by calculating the GHG emissions for each food product using the total CH_4, N_2O and CO_2 emissions associated with each sector, and taking the economic allocation of products into account, divided the total by the sectors total production. As the full life cycle of each product has not been comprehensively analysed it is difficult to know how accurate these results are. For example, 1kg of raw potatoes produced 49.2 kg CO_2e compared to only 1.7 kg CO_2e for a kg of potato flour, which would require more processing. This result for raw potato production was also 95% higher than presented by Williams et al. (2006) and Carlsson-Kanyama (1998). It is interesting to note that in the Netherlands study there is a large difference in the GHG emissions associated with individual plant protein sources. Apart from potatoes, rice, nuts and canned vegetables have much higher emissions than fresh vegetables and other grain options (Table 21). Carlsson-Kanyama (1998) also compared several plant protein sources and on a per-kg and per-gram of protein basis. The study showed that domestic, field produced potatoes, dry peas and carrots had lower GHG emissions than glasshouse produced tomatoes or imported rice produced in irrigated fields.

The storage, transport, cooking and food waste can have a large impact on the environmental performance of different food products. Jungbluth et al. (2000) found that the production of local organic meat had an environmental advantage over vegetables grown in a greenhouse, deep frozen vegetables, and potentially over vegetable products transported long distances by air. A Swedish study which evaluated food wastage generated by households found that approximately 52% of all vegetables/root vegetables bought by households were wasted after storage or food preparation compared to 10.6% of purchased meat/mince meat (Sonesson et al. 2005). This study highlights that food wastage can have a large environmental impact if the functional unit under study is the *kilograms of food consumed.*

This review highlights the lack of data available on the environmental impact of meat protein alternatives. To compare meat products to plant substitutes it is probably necessary compare entire meals rather than individual dietary components.

7.4.6 Comparative Studies – Meals and Diets

Because of the difficulty in comparing plant products directly to meat, most studies approach this by creating a ‘standard meal’ (i.e. a hamburger with a vegetarian patty vs. a meat patty) or by comparing a balanced diet based on nutritional needs. This is challenging however, because plant sources contain different proportions of digestible amino acids to meat. Consequently, a plant-based protein alternative to meat must contain a mixture of complementary plant protein sources to achieve the right balance of amino acids.

One recent Swedish study (Davis and Sonesson 2008a) compared the energy requirements and GHG emissions of four meals containing different proportions of grain legumes but equal proportions of protein and energy. The results showed the meal with a vegetarian burger had 50% less GHG emissions than a meal with a pork chop or a pork sausage containing 10% pea protein. However, the author acknowledges that the meals did not contain the same amino acid profile. Carlsson-Kanyama (1998) also compared the GHG emissions associated with two meat based and two vegetarian meals using different combinations of six locally sourced or exotic (imported or produced in a greenhouse) ingredients. Both the vegetarian meal and the pork-based meal made with locally sourced ingredients had lower emissions than the alternative pork and vegetarian meals that included exotic ingredients. This study highlights that production methods and the whole supply chain can influence whether a vegetarian meal is more environmentally friendly than a meat-based meal. Davis and Sonnesson (2008b) have also compared the GHG emissions associated with homemade and semi-prepared chicken meals using similar methods to Davis and Sonnesson (2008a). Interestingly the pork meals (Davis and Sonnesson 2008a) produce nearly 50% more emissions than the homemade chicken meal (Davis and Sonnesson 2008b).

Table 22 compares the results of these three studies that suggest on average a plant-based meal requires less GHG emissions than a meat-based meal, although there is a large range in both categories. All three studies compared complete meals containing the same quantities of energy and protein. Despite this, the plant-based meals are likely to contain less bio-available protein than the meat-based meals so they still are not directly comparable.

TABLE 22: COMPARISON OF GLOBAL WARMING POTENTIAL FROM PLANT-BASED AND MEAT-BASED MEALS OR DIETS FROM DATA PRESENTED IN THE LITERATURE

	Global Warming Potential		
	Average (kg CO_2-e / meal)	**Range (kg CO_2-e / meal)**	
Plant based meals	0.53	0.19	0.86
Animal based meals	0.93	0.64	1.15

Source: Davis and Sonnesson (2008a, b), Carlsson-Kanyama (1998)

Other studies however, have not shown the same result. For example, Wallén et al. (2004) compared the GHG emissions and energy use associated with the current average Swedish diet and a more nutritionally and environmentally sustainably diet, which recommended a reduction in meat, dairy products, sweets, soft drinks, dietary fats and rice and an increase in vegetables, fish, eggs, pulses and other cereals. They found a negligible difference in emissions; instead, they suggested that changing food production processes would have a greater in reducing GHG emissions than changing dietary patterns.

Although different protein sources may be comparable nutritionally, they may not be comparable in other areas that will determine their consumption, such as consumer satisfaction, availability and price. Generally, comparisons do not attempt to address these issues, because these decisions are largely driven by personal ethics rather than consumer satisfaction alone.

7.5 Implications of Shifting from Red Meat to Alternative Protein Sources

The literature indicates that shifting consumption patterns from eating red meat (beef and sheep meat) to pork or chicken will reduce the environmental burden associated greenhouse gas emissions, though the use of fossil fuels may in some instances increase. This is primarily driven by underlying differences in the digestive system, and the better feed conversion efficiencies for grain. However, GHG emissions are not the full story. Land use requirements and the implications of shifting from red meat to white meat or plant protein sources also should also be considered. These are likely to be covered by a parallel study to this report investigating biodiversity in the red meat industries.

7.5.1 Nutritional implications

As previously mentioned in section 7.4.4**Error! Reference source not found.**, different protein sources cannot be directly substituted for each other, as they are not nutritionally comparable. The nutritional value or quality of different proteins is governed by its source, composition of amino acids, rations of essential amino acids, its ability to be broken down during digestion and the effects of processing. The greater the ratio of indispensible amino acids in a protein, the greater the biological value or quality. Low-quality proteins are those that are deficient in one or more amino acids. For example, corn is limiting in tryptophan and lysine, cereals in lysine and soybean in methionine (Friedman 1996). Different meat protein sources have different amino acid compositions but are not deficient in any essential amino acids. Important differences exist between food products of animal and plant origin because of differences in the concentrations of essential amino acids that they contain. The concentration and quality of the protein in some plant-based food may be too low to make them the sole source of protein, but a well-balanced mixture of plant-protein sources is adequate. For example, a combination of soybean and a cereal should provide relatively adequate concentrations of all indispensible amino acids (Young & Pellet 1994). If the consumption of red meat is reduced, protein requirements can easily be met by substituting with white meat or plant protein sources. However, protein is only one dietary requirement. Adequate energy mineral and vitamin requirements also need to be considered.

7.5.2 Land and Energy Requirements

Few meat production LCA studies include land use as an environmental impact. To date no published data are available on the land use requirements for white and red meat production in Australia. However, Table 23 provides a summary of the few European studies that have included land use (Cederberg and Stadig 2003, Elferink and Nonhebel 2007, Weidema et al. 2008b, Williams et al. 2006). The production of 1 kg of beef in Europe requires nearly double the amount of land as lamb and nearly 4 times the amount of land required to produce 1 kg of pork or chicken. There is however a large amount of variation in land requirements within each source. The lower land requirement for pigs and chickens can largely be attributed to their higher feed conversion efficiencies i.e. to produce 1 kg of broiler chicken requires 1.7 kg of cereals, pigs require 2.35 kg and cattle between 5 and 10 kg cereals (Garnett 2009).

Land use studies have not often taken into account the *quality* of land required for the different species however, and a study based on both arable and non-arable land in Australia may favour red meat production. It is important to note that global availability of arable land is considered by some researchers as the greatest limitation to meeting future food demands.

TABLE 23: LAND USE REQUIREMENTS FOR ALTERNATIVE MEAT PROTEIN SOURCES FROM EUROPEAN LITERATURE

	Land Use Requirements		
	Average (m^2 /kg HSCW)	Range (m^2 / kg HSCW)	
Beef	42.3	23.0	58.9
Lamb	22.5	13.8	31.2
Pork	11.9	7.4	15.2
Chicken	10.2	6.4	13.7

The production of pork and chicken is more dependent on the use of cereal and oilseeds in their diet than ruminants. They therefore consume more grains that could be directly consumed by humans (Garnett 2009). The production of grain can only occur on arable land in areas of sufficient rainfall or under irrigation. As the world population grows, this land will increasingly be in demand for grain and horticulture to produce food for human consumption. Ruminants however, can be produced on non-arable land as they can digest sources of lower quality feed. Currently, most ruminant production systems supplementary feed with cereals as they provide a higher feed conversion ratio than grass and other low quality sources of feed and consumers prefer a grain-finished product. If ruminant production was restricted to non-arable land areas and cereals not supplemented, fewer ruminants could be produced and a greater volume of methane would be also be produced per kg of meat due to a reduction in feed conversion efficiency. On the up side, increases in methane production may be offset by reductions in other inputs including energy and water. Research is ongoing to find solutions that will reduce the volume of enteric methane produced by ruminants. Breeding animals with higher feed conversion efficiencies on lower-quality diets may also reduce livestock emissions.

Land use efficiencies could potentially be increased on arable land by adopting different management strategies. In lower rainfall regions, pasture cropping may be an option to increase both water and land use efficiencies. For example, in lower rainfall regions of Australia, most mixed–farming systems rotate pasture and cropping phases (i.e. only grow one cereal or pasture crop per year). Pasture cropping involves planting a cereal crop into an existing pasture and growing both simultaneously. This has several advantages. The pasture understorey suppresses weed growth, promotes the degradation of crop stubble, can be a source of leguminous nitrogen for the crop, and provides high quality feed for several grazings during the year. Additional benefits include the reduced need for nitrogen fertiliser for crop production, reduced reliance on chemical weed control as livestock help control weeds and possible soil health benefits from increased soil carbon accumulation from greater biomass production and breakdown.

The land use required for plant protein production is much less than animal protein. For example, soybean production requires 4.4 m^2/kg of product versus 7.7 m^2/kg chicken (Elferink & Nonhebel 2007). The land requirement for human cereal, oilseeds and other vegetable and fruit products is even less well documented in LCA studies than animal feed land requirements. This makes direct comparisons difficult. Srinivasan et al. (2006) examined what would happen to the production, consumption and trade of key commodities if OECD countries altered their diets to meat WHO/FAO nutritional guidelines. This would require a reduction in the consumption of vegetable oils, dairy products, sugar, animal fats and meat and a significant increase in the consumption of cereals, fruits and vegetables. The study found that the increased need to produce substitutes for animal products (cereals) only slightly outweighs the decline in feed cereal requirements i.e. the amount of cereals grown remains the same while the demand for animal products is drastically reduced.

Considering energy requirements, the literature indicates that beef production in Australia and NZ is less energy intensive that for northern hemisphere studies (13 MJ / kg compared to 60.6 MJ / kg beef). In an energy constrained world, this is an important result and highlights the role of the Australian red meat industries in producing low input protein products for the world. When these results are compared to European pork production, energy use for Australian beef is consistently lower, reflecting the requirements of pork for grain and housing, both of which are energy intensive.

7.5.3 Water and Land Interactions

Agricultural production in Australia is primarily limited by water availability and soil fertility. Red meat is largely produced on non-arable land and/or in regions of low rainfall, where it is the most suitable, or only suitable agricultural enterprise in most cases. Sheep and cattle have the unique ability to produce a valuable protein source off land that is unviable for other uses through their ability to convert low quality carbohydrate biomass into a high quality protein source. Because livestock can be economically managed at very low densities, they are able to harvest biomass from very low yielding regions. Because of this, animals are favoured in the world's most arid regions as the only means of yielding a useable product from the land.

It follows that if red meat production was to decline because of perceived environmental impact issues, alternative protein production from other animal species (pigs, chickens) or plant sources would not be suitable on the majority of this land.

Additionally, most oil seed and horticultural crops are grown under irrigation or in regions of higher rainfall. The impacts of climate change have already demonstrated that many current areas under crop production may not be viable in the future due to water shortages. This calls into question Australia's ability to significantly increase production of plant protein sources because of the competing pressures for good quality agricultural land and water availability.

There is further opportunity to increase the water use efficiencies of livestock and human cereal and horticultural production. Reductions in water requirements may be achieved by better matching plant water requirements when irrigating, improving soil water holding capacities, applying mulch to reduce soil water evaporation, breeding more drought resistant varieties and pasture cropping in low rainfall areas. Water is discussed in greater detail in chapter 9.

7.5.4 Economic Implications

In 2008 Australia had a total beef herd of 25.5 million head and a total sheep flock of 75 million head (wool and meat sheep). Production amounted to 2165 kt beef, 659 kt of sheep meat, 0.8 million head of live export cattle and 4.2 million head of live export sheep. The red meat industry directly contributed AUS$5435 million to the Australian economy from fresh, chilled and frozen meat (ABARE 2009). These figures do not include the value of by-products from red meat production including wool, leather, tallow, sheep skins, blood and bone or other by-products. Naturally it follows that a decrease in red meat production will result in a decline in supply of these by-products, and this should be taken into account when assessing overall environmental and economic impacts.

Red meat is produced across a wide range of land types in Australia, many of which are not suitable for other industries within agriculture or otherwise. This is particularly relevant for the large tracks of rangelands in semi-arid or arid zones. On the basis of GHG emissions or water usage per kilogram of meat, these regions would have the poorest performance because of the low reproductive rates and slow growth rates of animals raised in these areas. Consequently the industry is likely to suffer most in marginal regions where no replacement industries are likely to arise. The regional economic impacts for smaller country towns that rely on the grazing industries would be significant, and overall industry productivity would decline. The greatest effects would be felt in the northern beef industry, with knock on effects for live export trade, feedlot and grass-fed slaughter cattle supply. A full scale economic study has not been included in this review but has been addressed elsewhere by industry research.

7.6 Conclusions, Knowledge Gaps and Recommendations

7.6.1 Conclusions

A review of LCA research for red meat and alternative proteins suggests that Australia has the potential to produce an environmentally efficient protein product, with superior performance to many countries in the world. While not extensively supported by Australian research (there is only one LCA study to date for Australian red meat), there are underlying factors that substantiate this, such as the low intensity of energy use, and the low intensity of land based emissions, particularly nitrous oxide.

Many studies have presented meat, and particularly red meat as the 'worst' protein product from an environmental perspective, particularly with respect to greenhouse gases (i.e. Weidema et al. 2008b; Williams et al. 2006). This has led to calls for reduced consumption of meat products, particularly in Westernised countries.

The LCA literature indicates that red meat production generally produces more GHG per kilogram of product or per meal than white meat or plant protein alternatives. Most of these studies however do not allocate GHG emissions to animal by-products. It is noted that pork and chicken meat production result in less valuable by-products than sheep and cattle production (e.g. wool and leather). This makes the handling of co-products very important to the conclusions of comparison studies (Garnett 2009).

While there are some underlying issues related to red meat production that disadvantages these species in comparison to other animal species (i.e. the low breeding rate and ruminant digestion system). However, the comparative advantage of the ruminant digestive system (the ability to produce protein from low quality forages) is rarely taken into account. This ability means that, from a land use perspective, very few animal or plant products can be produced on the types of land that can be used for red meat production. Studies in Europe have not been well positioned to take this into account, as a high proportion of red meat production is reliant on grain and therefore arable land. Only one study (Williams et al. 2006) investigated two types of land to improve the assessment of grazing animals in their study, though the results were still skewed by the use of grains fed to livestock.

Another factor in comparisons is the reliance of different industries on energy. The international literature again shows red meat as energy intensive (i.e. Ogino et al. 2004; Weidema et al. 2008b). While some forms of red meat production (particularly those practiced overseas) are highly energy intensive, this does not need to be the case, as demonstrated by Peters et al. (2009a) for extensive, Australian red meat production.

The perspective on red meat will be driven by what is considered the most limiting resource or environmental factor. If greenhouse gas emissions are the limiting factor, red meat will need to make large gains in performance and is unlikely to be comparable to other meat products or plant alternatives. This may cause beef and sheep meat production to decrease where alternatives become more profitable (i.e. if carbon emissions are taxed), making the production of feedlot beef, for example, less competitive. While some land may be diverted to other agricultural production (such as grain production) most rangeland areas would not be suitable for any other form of agriculture and would most likely revert to low yielding wild-harvest systems with kangaroos and feral animals. This would lead to an overall reduction in food production and exports from Australia. This highlights the tension between the goals of food production and reduction of greenhouse emissions. A similar tension exists in the biofuels production debate between emissions reduction and land use (Wiedemann et al. 2008).

Because of the unique production practices carried out in Australia and our role in international trade, there is an opportunity for the red meat industries to substantiate their claim as being suppliers of comparatively low impact, resource efficient products. Considering our resources of non-arable, low rainfall land where alternative production options are limited, Australia could position itself to capitalise on these resources, in much the same way as New Zealand has done. This will rely on the promotion of Australian red meat as a resource efficient product from a broader perspective than greenhouse gas emissions alone however.

7.6.2 Knowledge Gaps and Recommendations

Knowledge gaps in this area include:

- Comprehensive LCA research for Australian red meat production that is representative of the major production regions across the country, and that investigates both average and 'best case' environmental performance based on up to date research.
- LCA research on the greenhouse gas emissions, energy usage and water usage associated with grain production in the eastern states where grain is used for feedlot beef production.
- LCA studies that compare 'like with like' for internationally traded meat products – i.e. Australian and US feedlot beef for the Japanese market.
- Improved on-farm assessment methodologies and emission factors for important factors such as enteric methane, manure emissions and land use N_2O.
- Improved impact categories to explain land use, highlighting the non-comparability of land used for red meat production or alternatives such as pork or dairy products (which rely on arable land) or plant proteins.
- Australian data for comparison of meat or alternative protein products in the publically available literature (these data are likely to become available from projects that are underway currently for meat chickens and eggs (completion date is the end of 2010), and from a recently completed project for pork).
- Further studies investigating plant proteins to reduce the variability in results currently available in the literature.
- Research on processed dairy products (particularly cheese) to enable a meaningful comparison with red meat.

To achieve this, the further research projects are required in the context of life cycle assessment. Recommendations are as follows:

- Data on the land use requirements for red meat production, including differentiation of grades of land suitable for various uses (at a minimum this should consider arable / non-arable land).
- Incorporation of vegetation / soils assessment (such as soil carbon sequestration) into life cycle assessment GHG calculations.
- Standardisation of important parameters for Australian LCA research (such as the handling of co-products at the point of slaughter). This could be achieved through a cross-industry project that can develop a simple user guideline to refine the current international standards that are available for LCA.
- Incorporation of scientifically rigorous sensitivity analysis within LCA to address key parameters that drive GHG emissions from red meat production, such as enteric methane, soil nitrous oxide and soil carbon sequestration. This should test the results of detailed research and allow the data to be presented with a 'whole of supply chain' perspective.

- Investigation of practical, realistic production alternatives that will alter GHG emissions and water usage, (i.e. herd manipulation, diet manipulation and inclusion of feedlots into the supply chain and others identified by industry and research – see recommendations from report 2).

The following specific studies may be useful to address gaps in the research:

- An LCA study may be commissioned to expand the number of beef and lamb supply chains investigated for impact categories of interest (i.e. GHG emissions, water usage, land usage etc). This study should consider both 'farm specific' data and national average data (such as data that can be supplied by the ABS / ABARE).
- A study may be commissioned to compare Australian beef and lamb production to other Australian meat products on factors that are of special relevance to red meat, i.e. land use and vegetation. This could be based on completed LCA projects (public reports are anticipated for pork, eggs and meat chickens by late 2010).
- A study may be commissioned to compare Australian beef production to international competitors (such as the USA and Brazil) within a study that uses the same methodology and parameters.
- A study may be commissioned to compare Australian lamb production to international competitors (such as NZ) within a study that uses the same methodology and parameters.

8 Water Usage in the Red Meat Industry

8.1 Introduction

Water scarcity is an issue of growing concern worldwide, largely because it is estimated that some 1.1 billion people do not have access to improved water supply sources (WHO 2009). With a growing human population, it follows that stress on water reserves will increase dramatically in the next 30-40 years (Rockstrom et al. 2007). While water scarcity is a relatively difficult term to define, there is little doubt that water resources are under considerable pressure worldwide (Shiklomanov 1998; Falkenmark et al. 1989; Glieck et al. 2009). An excellent review of these has been compiled by Rijsberman (2006).

At the simplest level, water scarcity is defined as the per capita per year water requirements for household, agricultural, industrial and energy supply sectors, and the needs of the environment. Falkenmark et al. (1989) developed a simple '*water stress index'* based on the above requirements and identified that 1700 m^3 of renewable water resources were required per person per year. Where supply falls below 1000 m^3 a country experiences *water scarcity*. It is of great importance to understand that the majority of this water requirement is used by agriculture in the production of food. Hence, most assessments of water scarcity do not relate directly to the supply of water available for domestic purposes (which are as low as 20 m^3/person/year – Rijsberman 2006) but rather provide an indication of the availability of water resources for food production worldwide. This is reasonable, as agriculture uses more water than any other activity in the world (Qadir et al. 2003).

Traditionally, water 'use' has been defined by water engineering terms and principles, and water use has only considered 'liquid' water. However, as researchers have sought new approaches to considering water scarcity, these traditional approaches have been broadened to include assessments of virtual water, which may incorporate embedded water derived from multiple sources. This alternative approach has introduced complexity and ambiguity to the term 'water use' and has created the need for new methodological approaches and clear documentation to avoid confusion.

In Australia, competition for water resources is one of the great challenges facing the most populated regions of the country. It has become increasingly apparent that water resources have been stretched beyond the sustainable limits for the Murray Darling Basin (MDB) where the majority of the population reside. As competition for water use grows, agricultural water use faces increasing scrutiny. This is not surprising, as agriculture accounts for 65-70% of water use nation-wide (ABS 2006). For these reasons water management is the subject of major state and federal political attention and funding, and has received increasing attention from the media and the community.

Water usage is highly relevant to the red meat industry for a number of reasons. Perhaps the most important of these, at a global level, is the effective use of water resources by agriculture (including red meat production) to meet the growing food needs of the world in an environmentally sustainable way. In the future, the red meat industry may also need to justify their water requirements and compete for secured entitlement of this resource. Australia's red meat industries (pasture fed beef and sheep, lot fed beef and lamb and the processing sector) are key users of water from the MDB, the Great Artesian Basin (GAB) and several other groundwater and river systems in Australia. Access and sustainable use of this resource is essential for ongoing operations, and security of water entitlements is particularly important for the lot feeding and meat processing sectors which often rely on regulated water supplies.

8.2 Water Policy Impacts on the Red Meat Industry

The functions of the water industry and the direction of water development are fundamentally dependent on water policy. Policy decisions seek to balance the allocations of water for various public and private benefits, and strongly influence the rights and abilities of individuals and corporations to access and use water.

Detailed national water policy in Australia has been developed over the past 14 years, beginning in 1994 with the Council of Australian Governments (COAG) water reform framework. This separated water rights from land - a necessary first step to expand trade in water. The reforms also sought to open up trading arrangements, including interstate trading. In 2004, COAG signed off on a policy blueprint to improve the way Australia manages its water resources - the National Water Initiative (NWI).

In the Australian Water Resources 2005 baseline study (National Water Commission 2005), states and territories estimated the current level of development in each water management area by comparing the current level of entitlements with the environmentally sustainable level of extraction (or sustainable yield). The study reported:

- of 340 surface water management areas, 3 (1%) were over allocated, and a further 44 (13%) were highly developed,
- of 367 groundwater management units, 19 (5%) were over allocated, and a further 85 (23%) were highly developed.

This is a surprising result considering the common perception that Australian water resources are highly stressed.

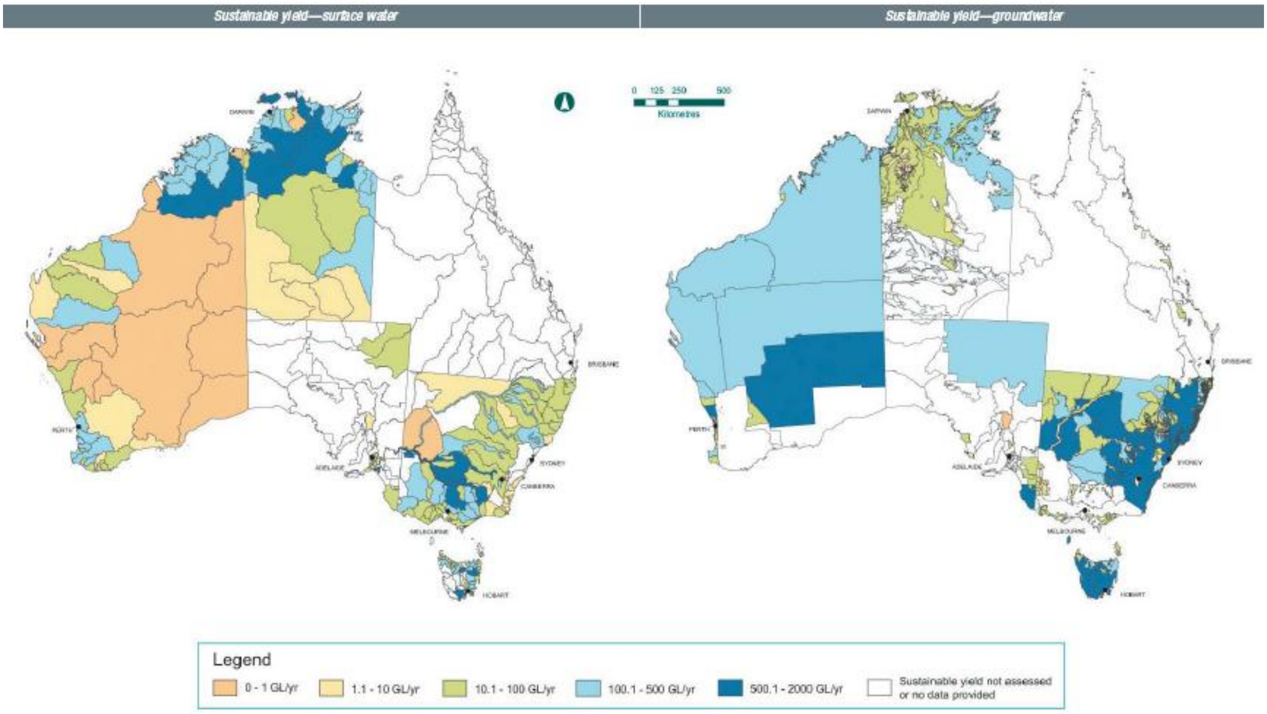

FIGURE 24: WATER RESOURCE ASSESSMENT FOR GROUNDWATER (RIGHT) AND SURFACE WATER (LEFT) – AUSTRALIAN WATER COMMISSION (2005)

Water policy across Australia is reasonably consistent, due mainly to the states and territories committing to NWI objectives for a nationally compatible water management framework. The objectives of the NWI – together with cooperation from the state and territory governments - will have the strongest influence on the direction of water policy in Australia for the near future.

Specifically, the NWI agreement aims to:

- facilitate and expand the trade of water and water markets,
- promote profitable and sustainable water use,
- reform the water industry to encourage confidence for investment,
- standardise and improve water planning and accounting,
- provide statutory direction to improvements in allocation of water for environmental and public benefit outcomes, and
- improve the management and efficiency of water in urban environments.

The promotion and implementation of water trading is possibly the most important of these objectives. Under the NWI, water trade is the transfer of water access entitlements (permanent) and seasonal water allocations (temporary) between different entities, for example, irrigators, intensive livestock producers, environmental water managers and water authorities (water infrastructure operators).

While the roll out of completed statutory water plans has been slow, the 2008 COAG update report on the water reform progress (National Water Commission 2008) found that almost all states and territories have made good progress in developing water access entitlement and planning frameworks as prescribed by the NWI, particularly in high-priority water systems. The NWI is due for a comprehensive review in 2010-11 to assess the extent to which its objectives have been achieved, review the performance of reform measures against performance indicators, and identify the impact of its implementation on regional, rural and urban communities.

Other policy initiatives similar to the NWI have been developed to build on the progress of its implementation, including the Intergovernmental Agreement on Murray-Darling Basin Reform and the Water for the Future plan. All of these initiatives aim to improve current water use inefficiencies, address over-allocations of water resources, promote the trade of water, plan for future demands for water caused by urban and rural development, and adapt to changing climate trends.

The administering authorities in each of the states and territories for rural and regional water policy are:

- NSW – Department of Water and Energy
- NT – Department of Natural Resources, Environment and the Arts
- QLD – Department of Environment and Resource Management
- SA – Department of Water, Land and Biodiversity Conservation
- TAS – Department of Primary Industries and Water
- VIC – Department of Sustainability and Environment
- WA – Department of Water

8.2.1 Extensive Grazing Sector

The consistency between states in regulatory reform and direction is reflected in most areas of water policy. The policies relevant to the red meat industry – primarily the entitlement to stock and domestic water - are no exception. The right to take and use water for extensive production of livestock is classified across the country as a fundamental right of property ownership. All states and territories have provisions for the use of both surface water and groundwater for stock and/or domestic purposes.

However, there are a number of variations on the allowable scale of works and the regulatory procedures that need to be followed. For example, the state of NSW makes provision for a landholder to capture and store a volume of surface runoff proportional to the area of their property to be used for any purpose. This entitles a landholder to the use of water equivalent to anywhere between 5 and 19 mm of runoff from their holding, depending on their location. In comparison, QLD landholders are entitled to intercept overland flow for stock and domestic purposes, but are limited to a total volume calculated from the carrying capacity of the property. The calculations allocate 20 m^3/hd for cattle and 4 m^3/hd for sheep, and also allow for climate factors specific to the proposed site.

Stock and domestic entitlements apply to groundwater resources as well as surface water resources. All landholders in NSW, VIC and TAS are required to obtain a permit to develop a bore for any purpose, although a license is not usually required for the use of the water. The same obligations to obtain a permit also apply to landholders in large portions of QLD, SA, WA and NT. Usually, the requirement to get regulatory consent coincides with highly developed or over-allocated aquifers.

It is reasonable to assume that the current relaxed standards of regulation and monitoring for stock and domestic water – from both surface water and groundwater sources - will come under review during the implementation of the NWI. With the stated objectives of addressing over-allocation and under-accounting, it is likely that a nation-wide commitment to licensing and metering will be required for stock and domestic water in the future.

8.2.2 Intensive Grazing Sector

The intensive grazing sector is defined as relying almost entirely on irrigation. Irrigation activity in Australia is highly regulated due to the pressures of limited water availability and high demand for the resource. The requirement to licence water extractions (from both surface water and groundwater sources) is universal across all states.

However, the availability of water for irrigation and opportunities for new irrigation licences are highly variable. In highly developed catchments such as the Murray-Darling Basin, the opportunities for irrigation are essentially restricted to existing licences and trading of entitlements. In less developed catchments such as the far north, new licences may be granted to currently unallocated water resources.

Each application for a new entitlement is assessed based on the proposed extraction conditions, existing users of the resource, and the availability of the resource. Existing licences are always given priority, and a new licence will not be granted if it will adversely impact on the performance of a downstream licensed user.

Because of the accessibility to water trading and the subsequent influence that this has on water value, it is likely that water use for intensive grazing of beef and sheep will diminish over time as users with a higher capacity to pay purchase water and / or properties with the express purpose of re-allocating the use of the water.

8.2.3 Intensive Livestock and Processing Sectors

The regulation and licensing requirements for intensive industries (including intensive livestock production and red meat processing facilities) are reasonably uniform across the country. All states and territories make the provision of a specific licence type for intensive livestock water extractions to cover water used in cattle and lamb feedlots. Abattoirs and processing facilities are generally classified as industrial water users and are licensed accordingly.

While current water resources for these sectors are protected by licensing, the pressure to compete for scarce resources is likely to become apparent in some catchments. Likewise, the potential to expand and construct new facilities will increasingly rely on water trading. Water trading would allow the feedlot industry to reallocate water resources over time in response to:

- changing commodity prices
- changing environmental conditions (e.g. salinity levels, river health)
- the changing availability of water.

For the feedlot industry, the key will be finding ongoing water access entitlements with high reliability. This will be most difficult in water management areas with over allocated or highly developed level of entitlements. The feedlot industry should be encouraging governments to look beyond the NWI commitment to develop 'compatible registers' of water entitlements to a common national registry system for water entitlements. This would achieve improved pricing and information disclosure and therefore transparency.

8.2.4 Non-Regulation Water Initiatives

A number of large scale water infrastructure projects are currently in progress with the aim of improving efficiencies of water transmission and water use.

The Great Artesian Basin Sustainability Initiative (GABSI) is a project in progress in Northern QLD with the aim of landholders – with funding assistance from Federal and State governments – capping artesian bores and replacing bore drains with pipe networks. After the first three years of GABSI (1999 – 2002), this resulted in an annual saving of 41,546 ML (Hassall & Associates 2003). The savings were achieved through the capping (rehabilitation or plugging) of 144 bores and the replacement of 3,541 km of open drains. This accounts for approximately 12 % of the eligible bores and approximately 15 % of total drain length. Targets for the whole project were set to control approximately 1,200 bores and replace approximately 23,000 km of drains in a 15 year period (ending in 2014).

A number of projects are also underway across Victoria to modernise irrigation and distribution systems, which are expected to deliver almost 400,000 ML of water savings annually. The water savings involved in the implementation of these projects will likely offset some of the impacts of reduced water availability in the short term.

The ***Wimmera Mallee Pipeline Project (WMPP)*** involves the replacement of approximately 17,500 km of existing open, earthen channels with pipelines. Landholders are initially provided with connections to the new pipe network free of charge, but are required to store the water in impervious tanks to maximise the water efficiency savings of the initiative. Other landholder responsibilities include the backfilling of channels and dams, and reconfiguration of their properties to take advantage of the new distribution system. The project aims to return approximately 87,000 ML of losses to the distribution network (GWM Water 2007).

Similarly, the ***Northern Victoria Irrigation Renewal Project (NVIRP)*** aims to return significant water savings through the replacement of inefficient water distribution infrastructure. Works include the automation of distribution schemes to meet water demands more efficiently and in a more timely fashion, the upgrading of metering equipment to minimise over-supply losses, and the rehabilitation of pipes and channels to minimise transmission losses. Incentives are also being offered to individual landholders for farm reconfigurations that make existing channels redundant, and there are approximately 18,000 farms currently eligible for review in the next 3 years. The project aims to return approximately 222,000 ML of system losses for urban supply, rural uses and environmental purposes (NVIRP 2009).

Smaller scale water saving projects are also being undertaken to improve efficiencies of stock and domestic water distribution across areas of southern NSW and Victoria. Small water supply schemes can suffer high water transmission losses in seasons where irrigation demand is low because open drains loose large volumes of water when filled initially as the soil takes up water. These systems often need to maintain the channels for users of water for stock and domestic purposes, and are exposed to considerable transmission losses to deliver these small volumes of water in some years. The Hay Private Irrigation District (PID) Pressurised Stock and Domestic Scheme in southern NSW is one example, saving around 1,000 ML of water annually by constructing a dedicated piped distribution network for stock and domestic water to service its 124 customers. A similar initiative is being investigated in the Wah Wah Water District near Griffith, with estimated savings of 10,000 ML annually (Water for Rivers 2009).

9 Water Usage Methodologies for the Red Meat Industry

With the current ambiguity and considerable range of values presented for 'water use' it is not clear how the industry can develop a pathway to improve the efficiency of water resource usage and decrease environmental impacts without clarification of the varied definitions of water use adopted by the different schools of thought. These perspectives have been developed for different purposes and may relate at some levels, but rarely all. The water use definitions found in the literature have been roughly grouped into three categories below:

- Water engineering framework: This 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, state). In Australia, the Bureau of Statistics (ABS) provide definitions for the consideration of water use, and engineers apply water balances to determine water use within a given system.

- Virtual water and water footprint frameworks: These approaches were initially designed to identify food products that require high levels of water for production. The objective being to reduce water stress in some regions / countries by importing products with high levels of embedded water rather than producing these goods locally. This concept has also been used as an indicator of environmental impacts from water use.

- Life cycle assessment framework: The LCA framework defines water in terms of i) volumetric resource use for a given product or service, and ii) the environmental impacts of resource use for a product or service.

The major variance in 'water use' figures for beef cattle quoted in the literature and media can be explained by the variance in definitions used and the boundary applied to the production system. The primary difference lies between the water engineering framework definitions and those applied in the calculation of virtual water.

9.1 Water Engineering Frameworks

9.1.1 Australian Bureau of Statistics (ABS) Definitions and Methodology

Definitions

While not a true water engineering definition, the ABS adopts an approach that is most closely aligned with this perspective. 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” if not. Essentially this definition corresponds to “Blue” water (liquid water that may be sourced from surface or groundwater supplies) and does not include rain falling on properties. For water to be considered “used”, it has either been transferred from its natural watercourse or extracted from groundwater. Hence, small overland flow dams used for watering livestock are not considered as water use.

“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*.

Methodology

The Water Account publications released by the ABS represent a collation of data from a wide range of sources. Water use statistics are derived from government agencies at all levels, water authorities, industry organisations, and a range of ABS surveys. It is reasonable to assume that organisations involved in the large-scale supply and transmission of water would base their information supplied to the ABS on metered data.

In cases where data has not been collected or where records are incomplete, values may be calculated or inferred from other related measures. An example relevant to agriculture is the volume of self-extracted water, where specific data does not exist due to monitoring impracticalities, and so the volume is inferred by subtracting the distributed water use from the total water use.

A similar water use accounting approach could be applied at scales right down to a farm level. Required data could be sourced from transaction records supplied by water suppliers, reports from government water authorities, and inferred calculations (such as calculating volumes from pumping rates and time spent pumping). The collation and analysis of data from these sources would allow a reasonably accurate assessment of water movements and usage on the farm.

More specific to agricultural production, the Water Use on Australian Farms publications present a higher resolution snapshot of water uses and sources. This includes a breakdown of irrigation activities by crop and method, as well as a breakdown of the sources of agricultural water (surface water, scheme supply, groundwater, reticulated mains etc.). The data presented in the Water Use of Australian Farms publications are derived principally through ABS surveys. Surveys generally ask respondents to provide areas of irrigation land and the volume applied to these areas. If the volume is unknown or unmetered, respondents are asked to estimate the applied volume, which in many cases would be inferred from an average crop water requirement value.

9.1.2 Water Use Estimates for Australian Red Meat Production following the ABS Definition

The ABS estimates that agricultural water usage in 2006-07 was 8,521 gigalitres (ABS 2008). Water use may vary year to year, and usage estimates for 2006-07 were around 25% lower than the previous year in response to decreased water availability as a result of ongoing drought conditions in many regions. The ABS does not estimate water use for the red meat industries directly, though some water usage categories accounted do relate to red meat production and could be used to generate broad scale water use estimates for the red meat industries.

Two research groups (Foran et al. 2005; Peters et al. 2009a) have estimated water use for red meat production using the definitions provided by the ABS (section 9.1.1). Additionally, Peters et al. (2009b) cites a study from the USA (Beckett & Oltjen 2003) which uses a similar approach. These estimates, together with estimates extrapolated directly from the ABS data are provided in this section.

Foran et al. (2005) estimated water use for a number of agricultural products based on their dollar value. Foran et al. (2005) did not include rainfall either as an input for feed consumption or where rain may collect on-site in creeks or dams. When cattle drink water that has been pumped or piped, this is included. Their methodology was based on an economic input-output analysis, which can be converted to L/kg of beef by using industry-level wholesale pricing - $3.5/kg beef and $4.2/kg (beef / sheep / pork / chicken products) calculated from their data. The masses refer to industry output of all products.

TABLE 24: WATER USE ESTIMATES CALCULATED BY FORAN ET AL. (2005)

		L/$	L/kg*
beef industry	direct	677	193
	total	731	209
meat products industry	direct	3.27	1
	total	333	79

* Estimated from these data using meat values

Peters et al. (2009a, b) followed a similar approach when estimating water use for Australian beef and sheep from three supply chains. This was done using the ABS definition of ‘water pumped’ and therefore excluded on-site water derived from rainfall.

TABLE 25: WATER USE ESTIMATES CALCULATED BY PETERS ET AL. (2009A)

Production system	**Victoria**		**WA**		**NSW**	
Production year	2002	2004	2002	2004	2002	2004
	Beef		Lamb		Beef	
ABS - water transferred from source	27	40	214	136	540	464

It can be seen from Table 25 that water use for beef production following the ABS definition may be very low. It should be noted that these estimates are not based on a direct calculation of drinking water and may in fact be lower than the drinking water requirement of the beef herd. For example, a basic estimate of drinking water for a cow-calf unit (including calf growth through to 12 months of age) would be as follows:

Drinking water / cow / year (40 L / day x 365 d) = 14,600 L
Drinking water / calf through to 12 months (20 L / day x 365 d) = 7,300 L

Assuming a 300 kg yearling and a 55% dressing percentage, the drinking water usage = 133 L / kg HSCW.

This does not contradict the values presented by Peters et al. (2009a) because there is no reason to include drinking water as a water 'use' unless the water is pumped from a source to a trough. Water consumed directly from farm dams and creeks is therefore excluded from the analysis.

To contextualise these results, Peters et al. (2009b) cite Beckett & Oltjen (1993) who have estimated water use for beef production in the USA using a roughly similar approach to the ABS definition. Water use was defined as "water which is diverted from possible use by humans" (Beckett & Oltjen 1993) and the analysis covered the entire national beef production system. The authors do not allocate between meat and slaughter by-products, but normalise water use on the basis of dressing percentages.

Peters et al. (2009b) re-evaluated the data from Beckett & Oltjen (1993) to enable comparison with Australian production systems by substituting irrigated feed and irrigated pasture with dryland conditions (as investigated in their study). Peters et al. (2009b) note that in the USA, 23% of the main feedstuff (alfalfa) is irrigated and there is a large (2 million ha) area of irrigated pasture. In contrast, only one of the Australian supply chains (the NSW beef supply chain) had irrigation for pasture production. Hence, Peters et al. (2009b) re-modelled the Beckett & Oltjen (1993) data removing inputs from irrigated feed, pasture and feedlot feed to provide a estimate that may be roughly compared with the Australian data (see Table 26).

TABLE 26: WATER USAGE ESTIMATES FOR USA BEEF PRODUCTION USING SIMILAR METHODS TO THE ABS

	USA water use estimate	per boneless beef product	With Aust. Grain data and without irrigation
units	ML/y	L / kg beef	L / kg beef
direct consumption	606,490	88	88
irrigated feed	4,311,977	627	7
irrigated pasture	11,242,607	1635	-
feedlot	153,288	22	22
feedlot feed	8,695,582	1264	-
meat processor	78,520	11	11
dairy calves input	236,645	34	34
Total	**25,325,109**	**3682**	**163**

Adapted from Peters et al. (2009b)

These results were broadly consistent to those reported by Peters et al. (2009a).

However, this comparison may be misleading at the broad scale, because Peters et al. (2009a) only investigate two beef production systems in their analysis, compared to the nationwide assessment of the USA completed by Beckett & Oltjen (1993).

For comparison, the following ABS water use data have been compiled and allocated to Australian national beef production in order to develop a first order assessment of the contribution of irrigation water to Australian beef production.

While the ABS does not measure water use for the red meat industries separately; several water usage categories are related to red meat production for feed inputs. These are presented in Figure 25.

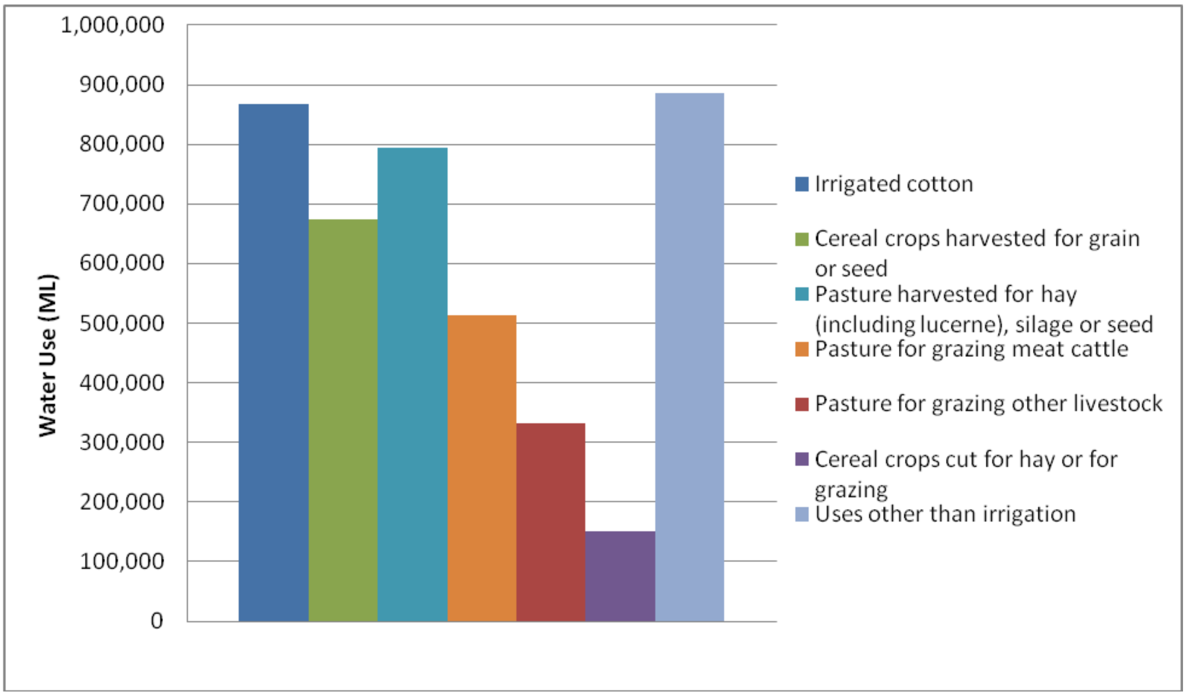

FIGURE 25: WATER REQUIREMENTS FOR A NUMBER OF COMMODITIES AND SECTORS RELATED TO THE PRODUCTION OF RED MEAT (ML) (ABS 2008)

The only category explicitly related to red meat production is 'pasture for grazing meat cattle'. However, irrigation water use for cotton (via cotton seed which is a major protein feed for the feedlot industry), cereal cropping (grain and forage) and 'uses other than agriculture' will all contribute some water use to red meat production. Water use such as pumped drinking water for extensive livestock production would fall under the category 'uses other than irrigation' as would water use for feedlots, though the proportion is not known.

To estimate water use from the Australian feedlot industry, Wiedemann et al. (2009) calculated water use using the estimate of 24 ML/1000 head capacity / year (QDPI&F – Skerman 2000) for the 2006 production year. Measured feedlot water use (Davis et al. 2008; Davis & Watts 2006) has shown consumption to be considerably lower than this, indicating that the QDPI&F estimate is conservative. Table 27 shows these estimated total water requirements for feedlot beef production across eight feedlot regions in 2006, based on the number of head-on-feed for this year.

TABLE 27: ESTIMATED FEEDLOT WATER REQUIREMENTS (ML/YR)

Region No	Region	2006
1	WA	2,369
3	Northern Territory	192
4	South Australia	709
5	Nth Queensland	2,981
6	Sth Queensland	11,242
7	Northern NSW	3,920
8	Southern NSW	4,852
9	Victoria and Tasmania	1,916
	Australia	28,120

Adapted from Wiedemann et al. 2009

Water used at feedlots would fall under the category of 'uses other than irrigation' and makes a small contribution to this category.

In an attempt to provide an approximate water use figure from these data, conservative 'best guess' allocations of water related to beef production via grain inputs (including cotton seed), irrigated pasture for hay (which includes lucerne) and 'uses other than irrigation' have been made for Australian beef production (see Table 28). These water usage data were then divided by the national beef production as reported by the ABS for 2006 (Table 28) to provide a value in L / kg HSCW. This approach is similar to Beckett & Oltjen (2003) study and is considered more representative than the estimate provided by Peters et al. (2009b) in Table 26.

TABLE 28: WATER USE CONTRIBUTION FOR BEEF PRODUCTION FROM IRRIGATED PASTURES, CROPS AND DRINKING WATER USING ABS DATA

	Australian water use (ABS 2008)	"Best guess" water allocation to the beef industry	Water (ML) allocated to the beef industry	L Water per kg beef
Water source	ML/y		ML/y	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.

It should be noted that the allocation of 'total water to total beef' will not be representative at the farm level. For example, most beef is produced on dryland areas with no water use from irrigation, while a small amount of beef is produced on irrigated land or using fodder from irrigated land. Beef produced on irrigated feed sources have an extremely high water use however. As a basic example, steers grazing one hectare of irrigated pasture may produce 400 kg HSCW / year, using 6 ML of water, amounting to 15,000 L / kg HSCW. When averaged over the whole beef herd this is an appreciable water burden for the industry. Despite this, the L/kg estimate using this approach is of a similar magnitude to both Foran et al. (2005) and Peters et al. (2009a).

As a summary, these approaches have been extrapolated to estimate total water usage for Australian beef production, using ABS 2006 beef production data (Table 29).

TABLE 29: WATER USAGE ESTIMATES FOR AUSTRALIA'S BEEF INDUSTRY

Literature source	Estimate (L / kg HSCW)	Total water usage (ML) from literature estimates and national beef production data (ABS 2008)
Foran et al. (2005)	209	449,609
Peters et al. (2009a) low	27	58,083
Peters et al. (2009a) high	540	1,161,668
'Best Guess' from ABS data	474	1,020,069

It is likely that average nationwide water use is closer to the higher estimate provided by the Peters et al. (2009a) study (540 L/kg HSCW). This is because of the substantial contribution from water use for irrigated pastures, which alone amounts to 238 L / kg HSCW when allocated across national beef production (Table 28).

As a ‘first order’ estimate based on the data in Table 29, water usage in the Australian beef industry is approximately 5-12% of the total ‘pumped’ water used by agriculture. However, further research is required to confirm this.

Meat Processing Sector

Water usage data are not reported separately by the meat processing sector, however various estimates can be sourced from the literature that utilise a similar definition to the ABS.

Meat processing plants are large users of municipal and/or bore water. Water use is driven by the necessity to maintain high hygiene standards to meet food safety requirements. A considerable amount of work has been undertaken into the measurement and benchmarking of water and energy use in the processing sector of the red meat industry in Australia. This work has led to the production of an eco-efficiency manual by Meat and Livestock Australia (Meat and Livestock Australia Ltd 2002) for the industry. This manual documents the resource use and waste generation data for a typical meat processing plant in Australia, as illustrated in Figure 9. These inputs and outputs are quantified in Table 4.

In red meat processing plants water is used for numerous purposes, including:

- livestock watering and washing
- truck washing
- washing of casings, offal and carcasses
- transport of certain by-products and wastes
- cleaning and sterilising of knives and equipment
- cleaning floors, work surfaces, equipment etc
- make-up water for boilers
- cooling of machinery (compressors, condensers etc.).

Surveys of water consumption per unit of production consistently show considerable variation. A factor that affects water consumption is cleaning practices. Plants which produce meat for export often have stricter hygiene requirements and therefore may consume more water for cleaning and sanitising (COWI Consulting 2000). Table 30 shows the breakdown (%) of water used in various areas of red meat processing plants based on a survey of industry.

TABLE 30: WATER CONSUMPTION IN AN ABATTOIR (MRC 1995 IN COWI CONSULTING 2000)

Major Areas of Water Consumption	Percent of Total Fresh Water Consumption
Stockyard (mostly wash down)	7 - 24%
Slaughter, evisceration	44 – 60%
Boning	5 - 10%
Inedible & edible offal processing	7 – 38%
Casings processing	9 – 20%

Rendering	2 – 8%
Chillers	2%
Boiler losses	1 – 4%
Amenities	2 – 5%

The slaughter and evisceration areas are the largest water users and responsible for the majority of cleaning and equipment sterilisation.

Water usage reported for the red meat industry by MRC in 1995 and MLA in 1998 (COWI Consulting 2000) ranged between 4 and 15 L/kg HSCW, with MLA (2002) reporting average values of 7 L/kg HSCW.

9.1.3 Farm Scale Water Engineering Definitions and Methodology

Definitions

The engineering approach to system water accounting describes water movements associated with system in the contexts of 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 a water use, delineation can be made between *beneficial* uses of water and *non-beneficial* uses, or losses. This is consistent with the approach used by the ABS in differentiating *consumption* from *use,* where 'beneficial' uses effectively correspond to consumptive activities.

With this clarification made, a more representative working definition of water use is the sum of beneficial uses. However, it is also understood that there are non-beneficial uses (losses) associated with beneficial uses, and these should also be included in the total water use value.

Methodology

The water engineering approach quantifies water use for a physical system through construction of a water balance. The technique is based on accounting for system inputs and outputs, with imbalances resulting in changes to system storage under the assumption that there are no net gains or losses (i.e. no water is generated or destroyed).

The strength of this approach – when used for water accounting – 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. At a farm level, typical water balance components are provided in Figure 26.

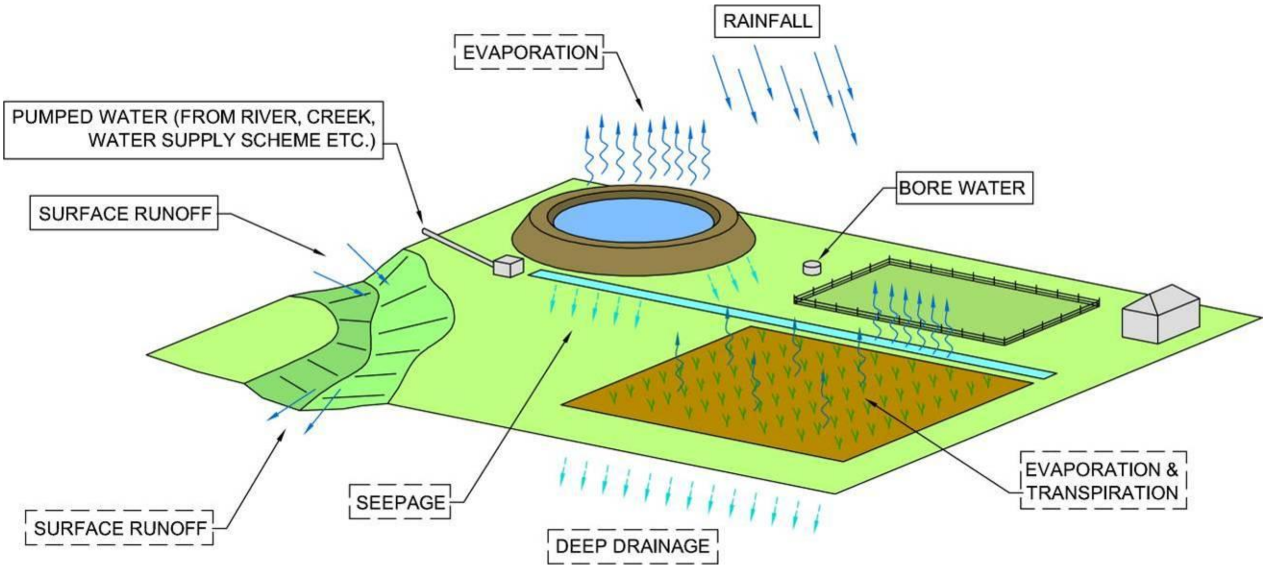

FIGURE 26: TYPICAL FARM WATER BALANCE COMPONENTS

The factors involved in this balance are a mix of physical processes and farm operations. The major components of a farm water balance are as follows:

Inflows – water may enter the system from many sources, which include:

- **Rainfall –** describes water entering the system through rainfall. This can be accounted for as direct input to storages or cropping areas, but can also indirectly account for the generation of surface runoff.
- **Pumped water –** describes water brought into the system via some form of pumping. Common water sources are rivers, creeks and bores. This can also account for water delivered to the property by some form of distribution network or water supply scheme. Measurements of these inputs are generally quite simple with the use of flow meters in pipelines and channels.
- **Surface runoff –** describes water entering the system while flowing over the ground surface. Surface runoff inputs are usually very difficult to quantify, except where they are transferred into storages or for direct use by pumps, pipes or channels.

Outflows – depending on the level of detail, the water balance will define outputs by measurement and deduction. Outputs include:

- **Transpiration** – describes the process of plants removing water from the ground to support life and growth, and the eventual release of that water as vapour to the atmosphere. Transpiration rates vary considerably between plants of different types and species, and also vary according to climate and environmental conditions.
- **Evaporation** – describes the loss of water from open water surfaces through vaporisation. The driving factors of the evaporation process are solar radiation, temperature, wind speed and humidity. Evaporation can be inferred from measured rates using a standardised pan, or calculated from measures of climatic conditions. Evaporation losses can be reduced by a number of approaches from engineering solutions (such as designing storages to minimise open water surface areas for the volume of water stored, covers for open water surfaces etc.) to management strategies (such as minimising time of storage, consolidating water into fewer storages etc.). In many locations across Australia, annual potential evaporation losses exceed annual

rainfall inputs. It follows that evaporation is a major loss component for many farm water balances.

- **Seepage –** describes outflow from storages and distribution channels by percolation through the base and/or walls. In engineered earthen storages or channels, seepage can be minimised through careful compaction of the lining material. The target minimum seepage rate from compacted earthen structures is approximately 0.000001 mm/day. Alternatively, seepage losses can be eliminated by the provision of a physical barrier, such as a plastic or concrete lining layer.
- **Deep drainage –** describes the infiltration of water into the ground beyond the root zone of plants. In the case of irrigation, deep drainage is usually triggered by applications of water in excess of what is required to fill the root zone of the planted crop to it maximum water holding capacity. Deep drainage can also be induced through rainfall onto a recently irrigated field.
- **Surface runoff –** describes water draining off the farm on the ground surface, usually triggered by rainfall. Surface runoff may be captured in storages, and can also include excess irrigation water draining off irrigated fields.

Water balances must also be applied to the individual components of a system to represent the behaviour of the components and describe the interactions between components. The generation of surface runoff requires analysis of a soil moisture balance, while water storages are also subject to a separate balance analysis to quantify fluctuations in storage volume.

If water use is to be attributed to production (i.e. L per kg of beef) the general approach would be to account for all 'system' water inputs (from watercourses, storages, groundwater etc) which are directly related to production. In this manner, rainfall is included in the balance, but is *generally excluded* from the calculations of 'water use' since it does not exist because of the production operation.

In the case of an irrigated crop, water use may be calculated as the water applied to the field without any further considerations of the water movements on the field after application. Depending on the level of available data, these water movements may be quantified to provide a better picture of the destination of the water if required (i.e. quantifying how much water ends up as deep drainage, evapotranspiration or runoff).

Farm water balance estimates are typically made using models of hydrology and crop production. A number of models are available for this purpose, including; PERFECT (Littleboy et al. 1996), HowLeaky? (Rattray et al. 2004), GRASP (Littleboy and McKeon (1997), CREAMS (Knisel 1980) and MEDLI (Gardner et al. 1996). A summary of these models is given below.

PERFECT

PERFECT was developed for cereal growing areas of the sub-tropics of Australia (Littleboy et al. 1996). It is a paddock-scale model that predicts the effects of: *climate, soil type, crop sequence and fallow management* on the water balance, erosion, and productivity. It was originally developed for research but is an extremely useful educational tool. The overall structure is physically based but individual processes may be represented by empirical relationships. Developed for sub-tropical grain growing areas of Queensland, the model has been successfully validated and applied to semi-arid areas of north Queensland and India.

HOW LEAKY?

HowLeaky? represents a rebuilding of the PERFECT model, with an enhanced interface designed to be useful to a range of non-modellers in exploring the implications of alternative land-uses on water balance, runoff, erosion and drainage. This is an experimental approach to find whether a more user-friendly interface will enable a wider range of users to use daily simulation models as an aid to land use planning.

HowLeaky? uses a simple leaf area index driven crop model (LAI model) and a generic pan evaporation model (ET: pan model) to represent pastures and trees. These models are responsive to water, temperature and radiation stress, and represent the dynamics between weather, soils and vegetation as these impact on water use and water and sediment flows. Since crop production is treated simply, these models should not be expected to simulate detailed crop management options such as soil fertility, detailed phrenology or population issues Rattray et al. (2004).

GRASP

GRASP is a deterministic, one-dimensional model of native pastures in semi-arid and tropical grasslands. It simulates the dynamics of pasture biomass and litter, the soil water balance and animal intake. The two main components of GRASP are the water balance and pasture growth sub models. A comprehensive description is provided in Littleboy and McKeon (1997). The soil water balance is calculated on a daily basis as the difference between inputs (rainfall) and outputs (runoff, drainage, soil evaporation and transpiration by grass and trees) in four soil layers of variable thickness and water holding characteristics. Runoff is calculated using the modified USDA Curve Number runoff model (Knisel 1980) and is a function of daily rainfall, antecedent soil water content and distribution, cover, and user specified runoff potential for bare soil and response to cover. This model is consistent with other agricultural water balance models such as PERFECT and APSIM (McCown et al. 1996).

The standard version of GRASP, which has a ‘free-draining’ algorithm, was used for most of the modelling. This version assumes that all soil water above field capacity will drain to the layer below in a single day, and therefore represents the maximum amount of water that could drain. A version of GRASP, which is currently under development, has the CREAMS drainage algorithm (Knisel 1980) from PERFECT.

CREAMS

The CREAMS hydrology model runs a continuous one-dimensional simulation of the water balance of the soil profile to the depth of the root zone (Knisel 1980). It is a generic model which allows the water balance of any crop to be partitioned into its components (soil evaporation, transpiration, soil water redistribution, deep drainage, infiltration and runoff). A daily time step is used, except in the optional Green and Ampt infiltration method.

Runoff is calculated using a modified USDA Curve Number (CN) method (Williams & LaSeur, 1976), with the potential retention parameter varied as a continuous function of antecedent available soil water. The form of the rainfall/runoff relation is similar to the *tanh* function used by Boughton (1966). Potential evaporation, soil evaporation and transpiration are calculated using the method of Ritchie (1972), using daily solar radiation and mean temperature, which are calculated from mean monthly values. Transpiration is determined using annual leaf area index (LAI) temporal patterns specified by the user. Root growth and water uptake distribution are simulated using the method of Williams & Hann (1978).

The soil is represented using seven layers, each being a proportion of the maximum rooting depth. The upper limit of plant available water capacity (UL), defined as total porosity minus wilting point, is specified for each layer. A parameter (FUL) is used to define the fraction of UL filled at field capacity, which is (1-FUL) is the proportion of UL that may drain. The same value of FUL is used for all soil layers. Soil water redistribution is calculated using a storage routing technique from Williams and Hann (Williams & Hann 1978). Drainage from a layer filled to above field capacity is a function of the volume in the store and an effective hydraulic conductivity parameter (RC).

A comprehensive description of the CREAMS model is provided in the CREAMS manual (Knisel 1980) and Williams & Nicks (1982).

MEDLI

MEDLI is a Windows® based computer model for designing and analysing effluent reuse systems using land irrigation, based on the PERFECT model. MEDLI requires daily time series climate data for estimating crop water requirements, simulating crop growth and carrying out water balance computations. The data required are rainfall, temperature, Class A pan evaporation and solar radiation (Gardner et al. 1996).

The waste estimation component of MEDLI generates, for a given industry, the daily composition and volume of effluent before pre-treatment, storage or irrigation. The simplest MEDLI waste estimation module uses measured waste stream details. Temporal variation in waste stream characteristics may be assigned monthly or seasonally, or for any other nominated periods, including single days.

The soil water movement is simulated as a one-dimensional (vertical) water balance, averaged over a field-sized area. The water balance component was taken from PERFECT (Littleboy et al. 1989; 1992) which was based on the Williams and LaSeur (1976) water balance models as used in CREAMS (Knisel 1980) and similar models. The calculation of plant available water holding capacity (PAWC) is determined as the difference between field capacity and the permanent wilting point. The method is an estimate only and is corrected by assessing restrictions such as potential rooting depth, sodicity, salinity and pH.

Soil runoff is predicted using the USDA Curve Number technique (USDA-SCS 1972) and is calculated as a function of daily rainfall, soil water deficit and plant total cover.

The plant growth module in MEDLI predicts the biomass accumulation and the quantities of N and P that are removed from the effluent irrigation site through crop growth and the export of harvested material. Flexibility is gained through the provision of a dynamic pasture growth model and a dynamic crop growth model. The pasture module is selected if a plant species is grown continuously, allowing regrowth to occur following mowing (rather than re-sowing the crop as occurs for the dynamic crop module). In this model, plant cover increases with thermal time according to a fixed sine-curve algorithm defined by the total thermal time to reach full cover. Growth is considered to be a function of solar radiation, plant cover and radiation use efficiency. Radiation use efficiency can be lowered by the highest of any stress due to temperature, water regime and low plant nitrogen. A diverse range of crop types is available in the model.

MEDLI contains a pond module (used in feedlot applications) which is a modified design model for treating pig wastes (Casey 1995). The module consists of mass balances for the hydraulic, nitrogen, phosphorus, potassium and total dissolved salts components. It uses a number of empirically derived relationships. The model allows for up to four effluent ponds in series. Nutrients in the incoming mass are partitioned between the sludge and the supernatant, and a transfer coefficient is used to estimate the nitrogen volatilisation from the pond surface. The pond module's function is to predict water levels and nutrient and salt concentrations. A nominated pond can be used for recycling purposes and the last pond may be used for irrigation.

MEDLI has the advantage of incorporating both standard water balancing program capabilities with specific modules that can provide detailed information for intensive livestock operations such as feedlots. MEDLI can also incorporate effluent nutrient modelling which is highly relevant for the feedlot and meat processing sectors.

9.1.4 Water Use Estimates Using the Farm Scale Water Balance Approach

As part of the MLA project COMP.094 – *Red meat production in Southern Australia – a Life cycle assessment* – a water modelling approach was adopted to develop water estimates for red meat production. The MEDLI model was selected to model the various scenarios involved in this project. Modelling was used to determine water movements in the plant system (rainfall, irrigation, evapotranspiration, leaching and runoff) and also to determine water movements at a feedlot. The modelling output was then used as an input to a property scale water balance. For all materials (including hay and grain) brought onto the property, the water used to grow the crop is included as an input to production. This water is primarily 'recycled water' under the definitions used in this assessment in the sense that it is supplied as rain and mostly returns to the local environment via evapotranspiration. Figure 27 shows a basic schematic of water flows on a grazing property as modelled in the project, showing the relative magnitude of water movements.

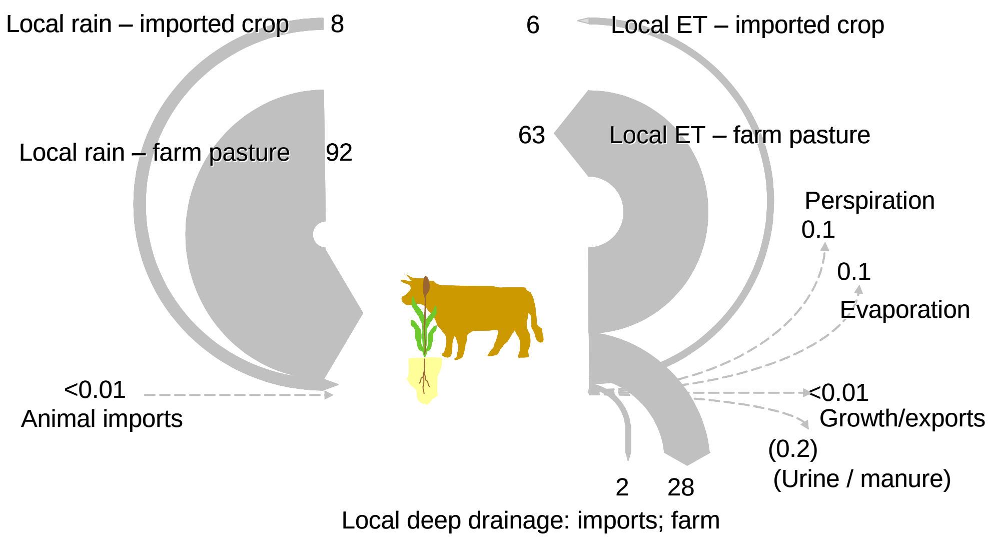

FIGURE 27: GRAZING SYSTEM WATER BALANCE FOR A BEEF OPERATION (PETERS ET AL. 2009B)

A more detailed summary of this work is found in the LCA review section (section 9.3.1).

Davis et al. (2008) used a water balance approach to quantify the water usage from individual activities within Australian feedlots. In their study, they developed and implemented a water monitoring framework at eight feedlots, which were selected to represent a cross section of geographical, climatic and feeding regimes within the Australian feedlot industry.

The framework suggested by Davis et al. (2008) included identifying water usage areas, developing a water resource flow diagram, identifying and installation of measurement tools, data collection and analysis. Figure 29 shows a basic schematic of water flows within a feedlot as described in the project. A mass balance approach was used to quantify water usage within individual components of the operation. Their approach also collected production data to integrate with the water balance data to allow standardisation of results for meaningful comparison over time and between feedlots. Integrating water usage data with production data is a critical step in any water balance approach.

The water usage for individual activities within Australian feedlots for Davis et al. (2008) is shown in Table 31. The variation, when standardised on a per-kg of HSCW gain, can be attributed to a number of factors. These include climatic variation, cattle genotype, cattle market types and management operations including frequency of trough cleaning, cattle washing, dust control and feed processing.

TABLE 31: WATER CONSUMPTION IN FEEDLOT ACTIVITIES (DAVIS ET AL. 2008)

Major Areas of Water Consumption	Water Usage L/kg HSCW gain*	Percent of Total Water Consumption
Drinking Water	22 – 86	78 – 91%
Feed Processing	0.4 – 2.4	1 – 6%
Cattle Washing	5 – 11.3	0 – 12%
Administration	0.6 – 3.2	0 – 5%
Sundry Uses	0.4 – 15	0 – 7%

* HSCW gain whilst in the feedlot.

Regulatory requirements imposed on feedlots ensure a correctly licensed, high-reliability water supply equivalent to 24 ML per year for each 1000 SCU of licensed capacity. Figure 28 expresses the total water usage data collected by Davis et al. (2008) on a megalitre per 1000 head-on-feed basis (Head is used rather than SCU for those states where SCU does not apply) from Davis et al. (2008).

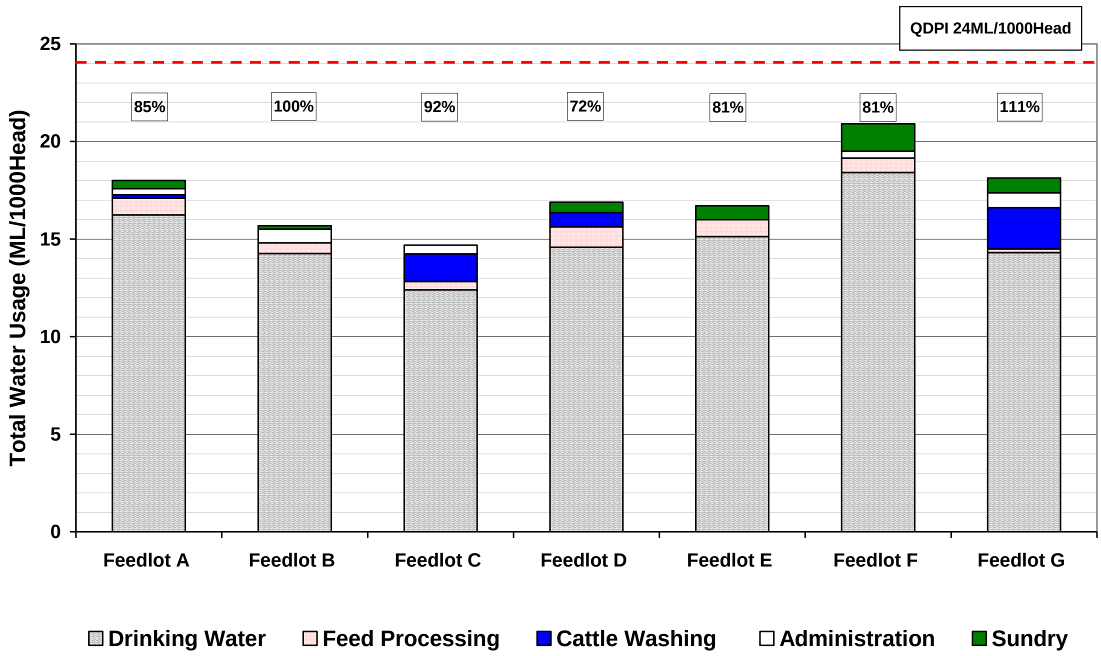

FIGURE 28: TOTAL WATER USAGE (ML/1000 HEAD-ON-FEED AND OCCUPANCY)

The data set published by Davis et al. (2008) puts valuable information in the hands of the industry to improve resource efficiency, meet the requirements of legislation and improve the sustainability of the industry in the face of a variable climate.

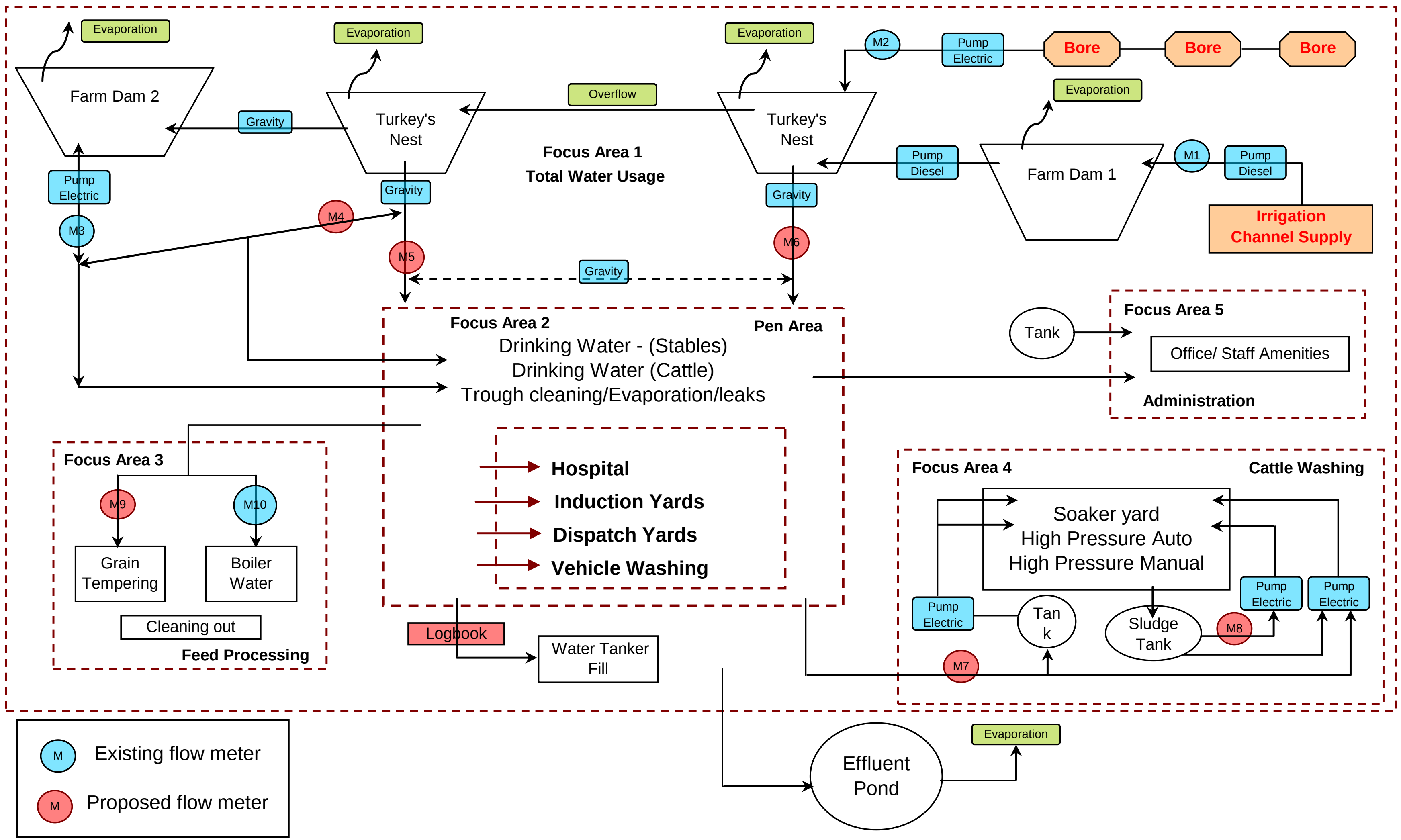

FIGURE 29: EXAMPLE FEEDLOT WATER SYSTEM FLOW DIAGRAM (DAVIS ET AL. 2008)

9.1.5 Catchment Scale Water Engineering Definitions and Methodology

Definitions

The application of a water balance approach varies with the scale of interest, and the definitions of inputs, uses and losses change accordingly. It follows that a catchment scale water balance model will account for water movements in a different manner to a farm scale model, despite the physical processes being identical.

An obvious example of this difference can be seen in the way that farm scale and catchment scale water balance models consider excess infiltration. At a farm scale, this mechanism is considered a loss (deep drainage), whereas the catchment scale model accounts for the mechanism as an increase in groundwater storage.

However, regardless of the scale differences the general engineering perspective remains the same in the sense that water use is considered to be the sum of the water outputs from a system, or the sum of the water inputs minus water captured in storage within the system.

Methodology

At the catchment scale, the water balance can be considered in much more general terms. However, the basic functions of the model and the water balance approach remain the same as those applied at a farm scale. Figure 30 shows a model which displays the factors involved in a catchment scale water balance, accounting for both surface water and groundwater resources.

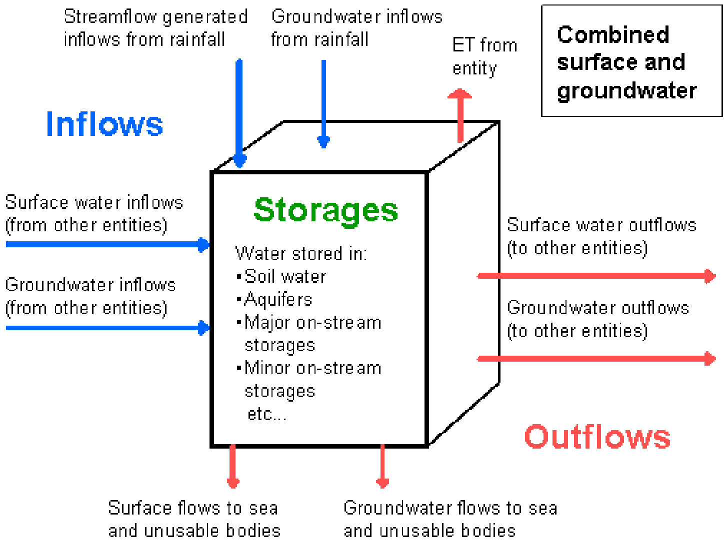

FIGURE 30: CATCHMENT SCALE WATER BALANCE DIAGRAM (NWC 2005)

While the terminology is different to the factors involved in the farm scale water balance, it is obvious that the inflow, outflow and storage functions are representative of the same hydrological processes. However, it is also obvious that a larger scale application of the water balance will result in a lower resolution analysis and will yield more general results.

9.1.6 Water Use Estimates Using the Catchment Scale Water Balance Approach

As part of the baselines studies undertaken at the beginning of the National Water Initiative in assessing national water resources, the National Water Commission produced a report detailing water availability in 51 priority areas. The analysis took a catchment water balance approach in assessing water availability by quantifying inflow and outflow components and examining the resulting storage conditions.

The water balance models used in this study varied in complexity and extent between catchments based on the availability of quantifiable inputs and outputs. In instances where measured data was not available or considered unreliable, modelling data or assumed parameters were used. In general, the inflows to the catchment water balance consisted of:

- Rainfall,
- Surface and groundwater inflows from other entities (upstream tributaries and aquifers outside of the catchment of interest),
- Transfers in (water pumped into the catchment from outside schemes), and
- Returns from the economy (water that has been delivered to a user and then passed on in the water cycle).

Outflows from the catchment water balances generally consisted of:

- Evapotranspiration (from storages, dams, wetlands, rivers, channels, springs and vegetation),
- Extractions and Diversions (water taken from sources, including any transmission losses),
- Stream and Aquifer outflows,
- Outflows to other entities (downstream catchments and aquifers outside of the catchment of interest), and
- Transfers out (water pumped out of the catchment into other areas).

Australian Water Resources 2005 (http://www.water.gov.au) 16/05/2007

WATER CYCLE REPORT Lachlan River - Regulated, New South Wales

Scope
Physical water entity: Lachlan River - Regulated
Jurisdiction: New South Wales
WMA ID: NSW_SW_412R
Period: 1 July 2004 to 30 June 2005

	Surface Water (ML)	Groundwater (ML)	Total (ML)	Data source	Data currency	Data accuracy	Method used to derive data (eg hydrographic, rainfall, etc)
Opening balance - water in store							
Major on-river reservoirs							
Wyangala Dam	107,900		107,900	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Carcoar Dam	2,500		2,500	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Jemalong weir	900		900	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Lake Brewster Weir	5,600		5,600	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Onstream minor and farm dams (unknown, record as zero - assume same at start and end of period)	0		0			E	
Major off-river storages (Major Farm Storages)							
Lake Cargelligo	13,400		13,400	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Lake Brewster	0		0	State Water Daily Operational Sheet-CAIRO	Aug-05	A	Obtain directly from sheets
Offstream minor catchment dams (unknown, record as zero - assume same at start and end of period)	0		0			E	
Renewable non-saline groundwater (in GMUs) (< 3500 mg/l)		70,366,000	70,366,000	DNR	Oct-06	C	Saturated thickness, water table drawdown depth, area and a drainable porosity. This is estimated active volume in storage.
Renewable non-saline groundwater (outside GMUs) (< 3500 mg/l)		0	0				
Renewable saline groundwater (usually zero)		6,330,000	6,330,000	DNR	Oct-06	C	Saturated thickness, water table drawdown depth, area and a drainable porosity. This is estimated active volume in storage.
Non-renewable groundwater (capable of being mined)		0	0				
Soil - unsaturated zone		0	0				
Snowpack (no snow)	0		0			N/A	No Snow
River channels (record as zero - assume same at start and end of period)	4,000		4,000	State Water	Sep-06	C	Estimated from CAIRO Sheets - Travel time times river flows and key river sites.
Unaccounted for storage (error item)			0				
Total opening balance	134,300	76,696,000	76,830,300				

Fate of Rainfall (not part of supply based water balance) Type of area	Area (km²)	Evapotranspiration (ML)	Rainfall (ML)	Deep Drainage (ML)	Runoff (ML)	Data source	Data currency	Method used to derive data
Forestry and plantations	6,590	1,811,652	1,820,767	24,824	2,751	BRS (Water 2010)	Sep-06	Modelled
Irrigated areas	437	151,150	153,297	4,010	209	BRS (Water 2010)	Sep-06	Modelled
Pasture	15,580	4,583,950	4,646,885	64,193	28,241	BRS (Water 2010)	Sep-06	Modelled
Dryland farming	3,419	1,191,512	1,214,687	51,885	1,578	BRS (Water 2010)	Sep-06	Modelled
Intensive use /urban	28	15,703	16,749	253	793	BRS (Water 2010)	Sep-06	Modelled
Bare ground	8	2,449	2,571	37	85	BRS (Water 2010)	Sep-06	Modelled
Water	207	74,713	77,149	1,111	1,416	BRS (Water 2010)	Sep-06	Modelled
Total	26,269	7,831,128	7,932,125	146,314	35,073			

	Surface Water (ML)	Groundwater (ML)	Total (ML)	Data source	Data currency	Data accuracy	Method used to derive data (eg hydrographic, rainfall, etc)
Rainfall to surface water runoff	35,073		35,073	BRS (Water 2010)	Sep-06		Modelled
Rainfall to groundwater recharge		146,314	146,314	BRS (Water 2010)	Sep-06		Modelled
Inflow to Surface Water							
Rainfall to surface water runoff (this is inflow in reservoirs, rivers, channels excl. baseflow) (see note 1)							
Storages (Wyangala, Carcoar)	108,000		108,000	State Water		A	Calculated from CAIRO Sheets
Rivers/Channels							
D/S Wyangala and Carcoar	118,000		118,000	DNR		B	IQQM Estimate
Rainfall/Runoff Harvesting	0		0	DNR		B	Assume minimal
Discharge from Groundwater to Surface Water (baseflow)	1,900		1,900	See Outflow to Groundwater section		D	
Returns from economy inside entity							
Urban treated effluent	0		0			E	no data available
Surface inflow from other entities							
Inter Valley Transfers	0		0	DNR		A	No intervalley Transfers
Returns from the economy outside of entity	0		0				
Unaccounted for inflow (error item)			0				
Inflow to Groundwater							
Recharge to groundwater (excluding irrigation)		115,000	115,000	Gross estimate (Partial Modelled - DNR)	Oct-06	C	
Recharge to groundwater from irrigation		1,600	1,600	Gross estimate (Partial Modelled - DNR)	Oct-06	C	
System gains							
Seepage from Surface water features (e.g dams, wetlands, etc)		1,700	1,700	Gross estimate (Partial Modelled - DNR)	Oct-06	C	
Conveyance losses (seepage from channels)		0	0				
Seepage from streams to groundwater		71,000	71,000	Gross estimate (Partial Modelled - DNR)	Oct-06	D	
Inflow from aquifers outside of entity (small - assume zero)		4,000	4,000	Gross estimate (Partial Modelled - DNR)	Oct-06	B	
Aquifer reinjection (e.g ASR)		0	0				
Unaccounted for inflow (error item)			0				
Total inflow to surface water and groundwater	227,900	193,300	421,200				

FIGURE 31: EXAMPLE WATER BALANCE ANALYSIS AT A CATCHMENT SCALE (KOLLMORGEN ET AL. 2007)

Assessment of these water balance components, combined with known storage levels at the beginning of the study, allowed catchment storage volumes to be analysed. Figure 31 shows a snapshot from a water balance analysis undertaken in this study.

The water balances established during this study were for the principal purpose of assessing the availability of water in key catchment areas, which was part of the greater objective of producing a national water account. However, the detail presented in the water balance can be used to assess and compare water movements and uses within the catchment. Detail is provided in the balance which describes the nature and extent of extractions to the economy, which describe water extracted for various commercial and urban uses. Estimates are also made on the losses associated with these uses.

Catchment scale water balance data such as these could be used to estimate broad scale water usage data for red meat production by comparing these with regional red meat production data such as that provided by the ABS through Agricultural Census. This approach has not been explored to date however.

9.2 Virtual Water and Water Footprint

Broad Definitions

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. The concept was proposed as an explanation of water stress alleviation in the Middle East / North Africa region. These regions have low volumes of water available for food production and manage this scarce resource by importing considerable quantities of food commodities rather than producing this food locally. This reduces the competition that agriculture may otherwise place on water resources, allowing greater supply for human (drinking, sanitation) and industrial purposes. The concept is based on the assumption that *irrigation* water is saved in the focus country through the importation of food. In the Middle East and North Africa this assumption is valid, because additional crop production could only be achieved through irrigation.

Hoekstra (2003) identifies two definitions of VW, i) the volume of water that was required to produce a product *in reality* (i.e. if wheat is 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), and 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. This concept expands on a traditional understanding of water use which is more commonly based on 'extracted water use' as is defined by the ABS for example.

To further improve the understanding of virtual water, 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'. These authors argue that a new paradigm is required in the description and consideration of water use and water use efficiency that incorporates green water into assessments of water resources and efficiency. The perspective taken by these authors is directed towards management of water scarcity in the face of growing demands for food production. Falkenmark (2003) contends that the food production needs of the world will increasingly require the optimisation of green water usage, because of the diminished and highly allocated reserves of blue water available world-wide.

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 red meat production based on the inclusion or exclusion of green water. This being said, few authors have made this distinction when presenting VW or water footprint data for meat production to date.

'Blue' and 'Green' Water Definitions:

Blue water represents liquid water most commonly considered under water usage estimations. Blue water usage may be derived from a variety of sources (i.e. groundwater aquifers, rivers, dams, lakes). This aligns closely with standard water usage definitions provided by water engineering.

Green water represents the evapotranspiration water requirements of a plant derived from the soil and prior to this, from rainfall only. It is also sometimes defined as 'soil stored moisture' derived from rainfall. This aligns closely to water use efficiency (WUE) for dryland crop production.

Further discussion of the vision and purpose of virtual water assessment can be found in Renault (2003).

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 it actually required to grow the given product, or an estimate of the avoided water, the water it would have taken to grow the crop in another country. This latter method is somewhat analogous to the concept of system expansion in life cycle assessment. Methodologies for the calculation of virtual water using both approaches have been reported by Hoekstra (Hoekstra & Hung 2002, 2005, Chapagain & Hoekstra (2003a, 2003b) and Renault (Renault 2003; Zimmer & Renault 2003).

These approaches are reviewed in this section as they relate to livestock. 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.

9.2.1 Water Footprint Definitions and Methodology

Water footprint studies have quantified global water trade in commodities associated with crop products (Hoekstra & Hung 2002, 2005) and livestock (Chapagain & Hoekstra 2003b).

Chapagain & Hoekstra (2003a) define virtual water as the *"volume of water required to produce a commodity or service"*. This is a country specific definition and is primarily aimed at determining international flows of virtual water as a means of alleviating water stress. Virtual water flows between nations have been completed by these authors for trade in crop products (Hoekstra & Hung 2002) and livestock products (Chapagain & Hoekstra 2003b). Depending on the region in which the product is grown and the efficiency of production, the water footprint of a product may vary greatly, though meat products are naturally more 'water intensive' than crop products

(Hoekstra & Chapagain 2007). The methodology for estimating the water footprint of livestock products is reviewed here.

Chapagain & Hoekstra (2003a) define virtual water of livestock products as the sum of the following water uses *over the lifespan of the animal:*

- Water used to grow and process feed,
- Drinking water,
- Cleaning / sundry water,
- Process water (i.e. meat processing).

From this definition, it appears that water used in upstream processes (such as breeding) is not included (see Figure 32). Essentially this equates to a system boundary (to use LCA terminology) at the farm boundary. Water usage from other inputs to production such as energy is not included.

The water required for feed production is modelled for each feed crop type, using the average specific water demand for that crop at a country specific level. The virtual water content of each crop is calculated using the method described by Hoekstra & Hung (2002). This is based on crop water requirements, which are estimated using the CROPWAT model developed by the FAO. This is a retrospective, desktop calculation based on broad assumptions with no specific measure of water source or water loss pathways.

Likewise, drinking and service water are estimated from data sourced from the literature, which is of variable quality.

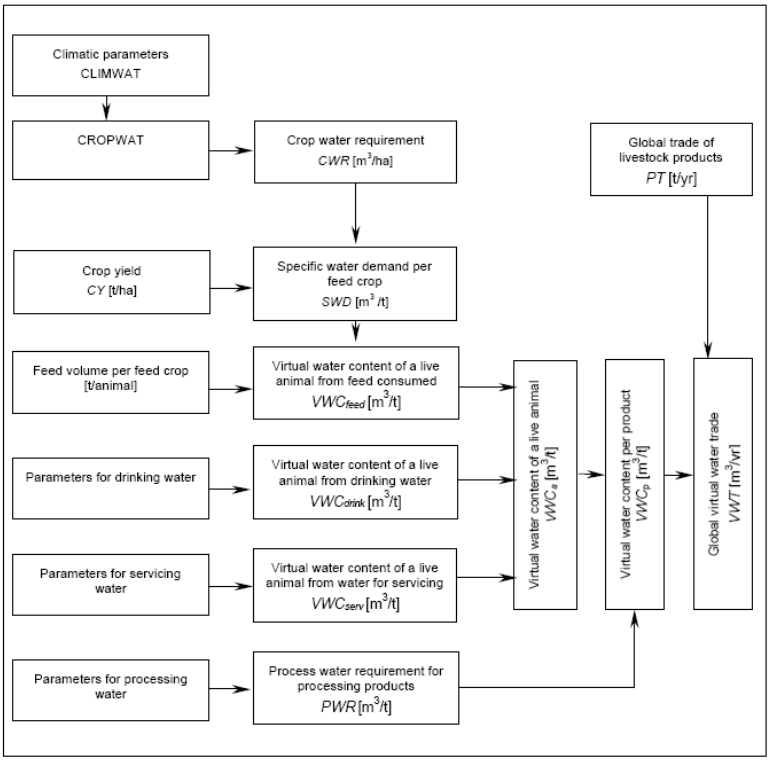

FIGURE 32: STEPS IN THE CALCULATION OF VIRTUAL WATER WITH INTERNATIONAL LIVESTOCK TRADE (CHAPAGAIN & HOEKSTRA 2003)

To identify the role green water plays in the water footprinting methodology, Peters et al. (2009) re-analysed water usage data from Hoekstra & Chapagain (2007) to remove the rainwater component from the production of grains. Hoekstra & Chapagain (2007) cite 1334 kL water is required to produce one tonne of wheat. For comparison, Narayanaswamy et al. (2004) report 0.6 kL/t for Australian dryland wheat products (process water only, excludes rainfall).

The water demand for Australian and global beef production from Hoekstra & Chapagain (2007) is presented in an adapted form in Table 32. Water use for Australian production was not broken down into the components shown for global beef production. However, by substituting the data for dryland wheat into the global estimate, the effect of rainfall in feed production can be removed.

TABLE 32: WATER USAGE ESTIMATES FOR BEEF PRODUCTION USING GLOBAL AND AUSTRALIAN DATA

	Hoekstra & Chapagain (2007)		**Peters et al. (2009b)**
	Aust. estimate	**Global average**	**With Aust. grain data**
Units	L / kg beef	L / kg beef	L / kg beef
Direct consumption	-	155	155
Feed	-	15,340	***7***
Total	17,112	15,497	162

Source: Reproduced from Peters et al. (2009b).

For an Australian example, this shows the relative contribution of blue and green water to red meat production in rain fed operations.

Once the virtual water use has been calculated for the live animal (or a quantity of product – i.e. 1 tonne), an allocation process is applied to proportion water use to primary products at the point of slaughter (i.e. beef carcass, hide, offal). This is done using an economic allocation process, by multiplying the mass of water used for producing the live animal by the value fraction of each component, divided by the mass fraction of each component. This can be seen in an example for Canadian beef below, taken from Chapagain & Hoekstra (2003a).

Example 1 – Water use allocation between components of a live bovine. Initial water use for the live animal is 9629 m^3 / t and comprises 9619 m^3 / t of VWC and 10 m^3 /t of PWR. Economic fraction for the carcass is 0.802, the mass of carcass is 0.6 t.

$$VWC[Canada, carcass] = \left[\frac{(9629+10)\times 0.802]}{0.60}\right] = 12864 \text{ m}^3/\text{t}$$

$$VWC[Canada, offal] = \left[\frac{(9629+10)\times 0.128]}{0.15}\right] = 8199 \text{ m}^3/\text{t}$$

$$VWC[Canada, skin] = \left[\frac{(9629+10)\times 0.071]}{0.08}\right] = 8513 \text{ m}^3/\text{t}$$

A more detailed breakdown of the economic allocation process for Canadian beef is shown in Figure 33 below.

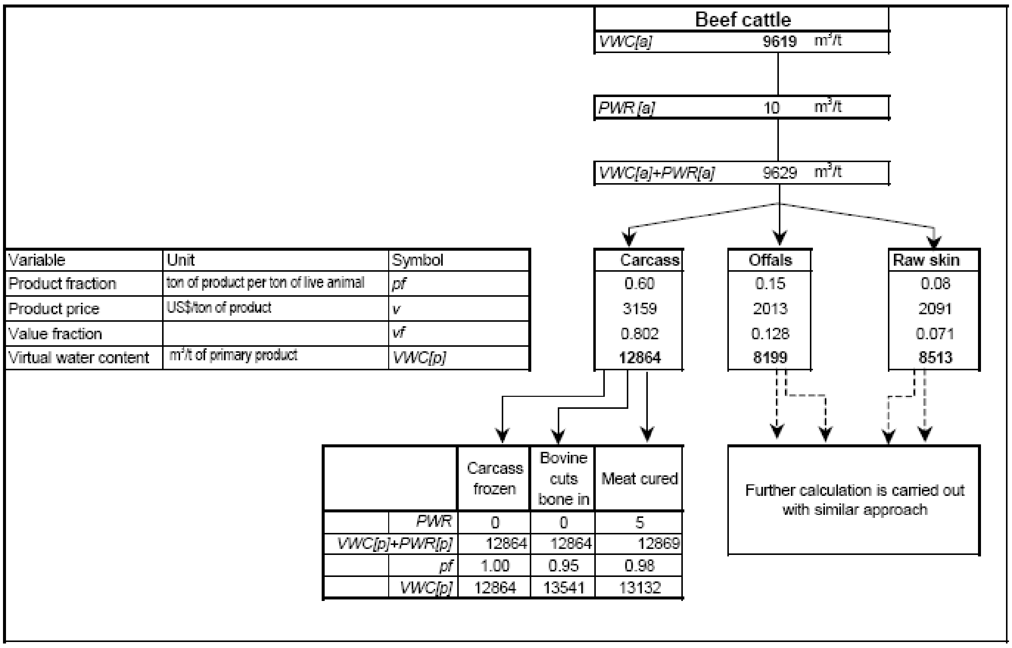

FIGURE 33: ALLOCATION OF VIRTUAL WATER TO CO-PRODUCTS FOR BEEF (CHAPAGAIN & HOEKSTRA 2003)

At a nation-wide level, three systems of livestock production are modelled. These are; grazing systems, mixed grazing and grain fed systems and industrial systems (housed, grain fed). Virtual water use is estimated for each of these systems separately, and the overall results for a nation are apportioned based on the relative percentage of each system in that country.

Chapagain & Hoekstra (2003a) do not incorporate a methodology for distinguishing between blue and green water, or differentiating water by source. Hence, the usefulness of the results for drawing environmental impact implications are limited.

The points of consensus and divergence of this approach from other virtual water calculation approaches, and from other methodologies, will be discussed in the summary sections.

Chapagain et al. (2006) also incorporates dilution water into the water requirements of cotton production, in addition to water required to grow and process the crop. This has been calculated by determining the volume of emissions from agriculture (primarily nitrate) and determining the water required to ensure the concentration of nitrate does not exceed the levels set for human health (10mg/L). Following this method would require careful assessment in livestock systems, where concentrated effluent is commonly used on-site. In this case, 'emissions' would only represent those leaving the system in surface or groundwater.

9.2.2 Virtual Water – Renault

A second calculation method for VW has been presented in Renault (2003) and Zimmer & Renault (2003). Renault (2003) defines virtual water as an assessment of the "***water that is or that would be consumed***" in producing a product. This definition therefore expands the approach to consider production in alternative locations, or even production of alternative products, in the assessment of virtual water. Renault also presents a *demand driven* perspective on water use, suggesting that the composition of diets is a primary driver of global water demand and that water use can be reduced through reduction of animal based products in the diet (i.e. Renault & Wallender 2000).

Renault (2003) defines several principles that explain different perspectives and methodologies for virtual water assessments, these are:

- 'Standard values' for VW use based on production in the dominant production regions of the world, i.e. 'real water use'.
- 'Marginal water consumption' for VW use based on the amount of water that the product *would have required if grown in the location of interest*.
- 'Nutritional equivalence', or the water required to produce a food product with a set of nutritional properties (a set balance of protein, energy etc).
- 'Substitution' VW use, which represents the amount of water that it would require to produce a similar product that can be substituted for the product of interest.

These new definitions introduce a greater degree of complexity to an assessment of VW.

For calculating **standard VW values**, Zimmer & Renault (2003) propose a considerably more detailed assessment of water use at the field level. The assessment is based on the water accounting (water balance) approach, and the VW for the crop product is to include both evapotranspiration and losses (i.e. seepage and deep drainage) that are not recycled at the basin level.

In cases where the '**marginal VW**' is calculated, these results will not be comparable from country to country. For example, the marginal VW for wheat imported to the Middle East will be different to the marginal VW for importing wheat to Europe, where grain production requires less water per kg of yield.

Substitution and Nutritional equivalence VW is used to calculate the VW of a by-product or a product that does not use water in its production based on the production of a nutritionally equivalent product. This allows for the estimation of VW for products that take no water to produce at all, such as fish caught from the ocean, by assessing the fresh water requirements *that would be necessary to replace these products with alternative animal products*. For a by-product, the substitution VW is the water it would have taken to produce a nutritionally equivalent primary product.

As a consequence of these definitional and methodological differences, estimates of VW for *exported* grain products made by Zimmer & Renault (2003) are significantly higher and less precise than those presented by Hoekstra & Hung (2002, 2005). For *imported* products (where the marginal VW is calculated), the results are not comparable with Hoekstra & Hung (2002, 2005) as they depend not on 'real water use' values, but avoided water use in the country of interest.

9.2.3 Virtual Water – Pimentel

Though the term is not used by the author himself, Pimentel (Pimentel et al. 1997; Pimentel & Pimentel 2003; Pimentel et al. 2004) is a prolific author of water usage estimates for animal products. No clear methodology could be determined from these sources; however the following general distinctions can be made between estimates from this author and other VW assessments.

Pimentel estimates water usage as the sum of feed requirements and drinking requirements of livestock. However, his estimates for water used to produce feed for beef and sheep are considerably higher than other VW assessments. It is possible that the difference comes from using average precipitation to grow forage and grain crops rather than modelled evapotranspiration. For example, Pimentel et al. (2004) quote the following requirements to produce 1 kilogram of beef from rangeland systems in the USA:

"If cattle are raised on open rangeland and not in confined feedlot production, 120 to 200 kg forage is required to produce 1 kg beef. This amount of forage requires 120,000 to 200,000 L water per kg, or a minimum of 200 mm rainfall per year."... Pimentel et al. (2004).

It is noted that water requirements for grain fed beef in this same paper are considerably lower (43,000 L) indicating the difference in efficiency in the two systems. From the quote above, it is assumed that water requirements for pasture production are equated directly with rain falling on the pasture land. Considering the very high pasture intake figures provided per kilogram of production, and from other comments in the paper, it appears that Pimentel includes the water requirements of the whole herd used to produce slaughter cattle.

Pimentel et al. (2004) does not differentiate the source of water, and appears to take a definition that is broader than any taken previously, contributing to the very high estimates provided from this author. This author presents conclusions linking VW water use with river flows in the USA which are difficult to substantiate given the estimation approaches taken. Elsewhere, the same author uses these data as a basis for recommending a reduction in meat consumption by the public (Pimentel & Pimentel 2003).

9.2.4 Virtual Water Usage Estimates for Red Meat

Many assessments of water usage for red meat production have been made using the virtual water and water footprint methodologies. Not surprisingly, these estimates vary greatly (see Table 33).

TABLE 33: LITERATURE ESTIMATES OF VIRTUAL WATER REQUIRED TO PRODUCE ONE KILOGRAM OF BEEF

Water Required L/kg beef	Functional Unit and Boundary	Inclusion of Blue / Green water	Research location	Reference
120,000-200,000	Unclear – Pasture fed cattle, likely to include upstream impacts from breeding	Includes both without distinction	USA	Pimentel et al. 2004
105,400	Unclear – Pasture and grain fed cattle, likely to include upstream impacts from breeding	Includes both without distinction	USA	Pimentel et al. 1997
15,000 – 70,000	1 kilogram of meat, Boundaries are unclear	Includes both without distinction	not known	Gleick, in Gleick et al. 2009
43,000	Unclear – Grain fed cattle, likely to include upstream impacts from breeding	Includes both without distinction	USA	Pimentel et al. 2004
17,112	Boneless beef [1]	Includes both without distinction	Australian average	Hoekstra & Chapagain 2007
15,497	Boneless beef [1]	Includes both without distinction	World average	Hoekstra & Chapagain 2007

[1] Water use is over the slaughter animal's lifetime only and does not include upstream impacts

TABLE 34: LITERATURE ESTIMATES OF VIRTUAL WATER REQUIRED TO PRODUCE ONE KG OF SHEEP MEAT

Water Required L/kg sheep meat	Functional Unit and Boundary	Inclusion of Blue / Green water	Research location	Reference
51,000	Unclear – Pasture and grain diet, likely to include upstream impacts from breeding	Includes both without distinction	USA	Pimentel et al. 2004
6,947	Boneless sheep meat [1]	Includes both without distinction	Australian average	Hoekstra & Chapagain 2007
6,143	Boneless sheep meat [1]	Includes both without distinction	World average	Hoekstra & Chapagain 2007

[1] Water use is over the slaughter animal's lifetime only and does not include upstream impacts

A rapid re-analysis of the Australian red meat LCA water usage data suggests that as little as 2% of the total virtual water used to produce beef in Australia is derived from blue water, with the balance being sourced from green water.

From a VW or water footprint perspective, meat is a more 'water intensive' product than a plant product (Chapagain & Hoekstra 2007). This has been used as an argument to reduce meat consumption (Renault & Wallender 2000). However, without knowing anything about the form of water used (blue or green), the land used in the production of the product (arable or non-arable) or other contributing factors, it is impossible to state that reducing consumption will result in genuine water savings. Despite the obvious problems in interpreting these data for livestock production, differentiation of water type between blue and green sources is rarely presented in the literature or the methodologies for calculating VW or water footprints. This is a flaw in the methodologies proposed, particularly for nations like Australia that rely heavily on rangeland beef production on non-arable land, where green water use is high and water has a low degree of transferability to use with other products.

In addition to the contribution that VW and the water footprint concepts make to understanding water scarcity and food production, Hoekstra (2003) identifies the concept as an indicator of the environmental impacts of a producing a given product. According to these authors, environmental impact is implied by the magnitude of the water footprint (Hoekstra 2003; Hoekstra & Chapagain 2007). This application of the concept, particularly when differentiation of the source of water (blue or green) has not been reported, is partially useful at best, or misleading and damaging at worst. Few authors of VW estimates have been willing to elaborate on the environmental impacts of green water use for red meat production. This is a complex issue that is integrally related to land use and land capability. A more accurate assessment will incorporate land use factors that differentiate land capability between arable and non-arable land.

We contend that the virtual water and water footprint concepts in their current form are not able to provide adequate detail to be of value in environmental assessments of water usage in red meat production.

As a trade tool for alleviating water stress by trading ‘embedded’ water with products, the virtual water concept has merit. However, as a proxy for the environmental impact that water usage has on aquatic environments (i.e. rivers), the concept is misleading when no differentiation of the source of water (blue or green) has been clearly elaborated and correctly interpreted in the results and discussion.

9.3 Life Cycle Assessment

9.3.1 Water Usage Definitions and Methodology

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. 2008). This being said, a number of methods for the assessment of freshwater use have now been proposed, and several options are sufficiently developed to be compared with alternative methods for water accounting.

LCA has 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).

Water Quantity Indicators and Use Types - Owens

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 and 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 unreplenished source. Owens represents one of the founding methodologies presented 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.

Freshwater Ecosystem Impact and Fresh Water Depletion – Mila I Canals

Mila i Canals et al. (2008) have expanded and modified the approach provided by Owens (2002) to provide water characterisation factors for freshwater use. Mila i Canals et al. (2008) 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. (2008) 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.

In their development of impact categories, Mila i Canals et al. (2008) identify two main aspects of water that need to be considered, i) water as a resource, 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 (2008) suggest omitting this aspect from LCA.

Changes of water quantity affecting ecosystem health (FEI). Water supply for ecosystem function is under pressure in many regions of the world, and needs to be assessed under any comprehensive LCA method. Mila i Canals et al. (2008) suggest the FEI impact category to assess this. The primary driver of FEI will be abstractions for evaporative uses of water upstream, as these remove water from the system thus depriving the ecosystem. Mila i Canals et al. (2008) suggest that abstractions from groundwater sources are not considered under this impact category as they would not have contributed or taken from ecosystem function prior to human intervention. This may not be the case in some sub-artesian basins of Australia, where surface and groundwater sources are closely linked. Green water use is also omitted from this impact category. Water quality impacts are not covered by this indicator, but are left for assessment under other impact categories.

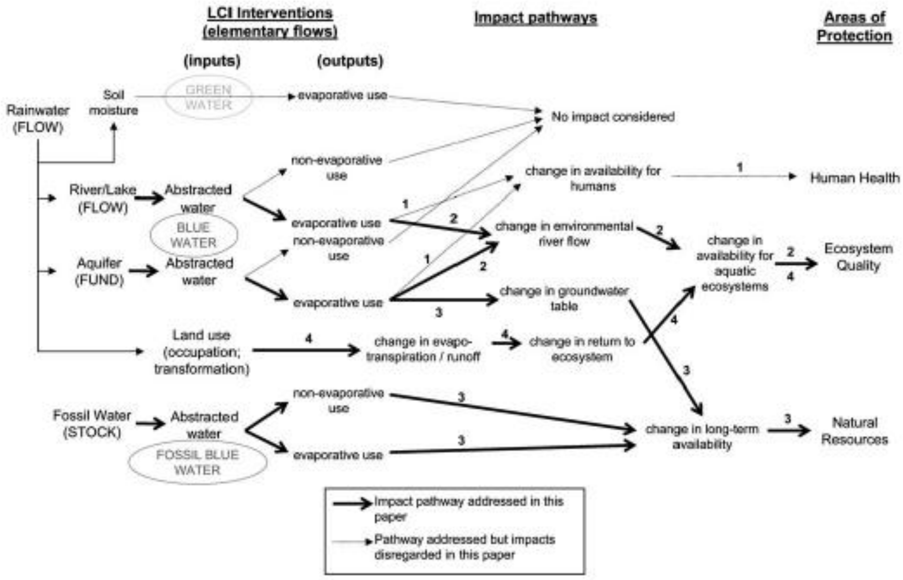

FIGURE 34: MAIN IMPACT PATHWAYS RELATED TO FRESHWATER USE (MILA I CANALS ET AL. 2008).

Depletion of freshwater resources (FD). This impact category has been developed to assess water used from stocks (i.e. groundwater). Mila I Canals et al. suggest that use of water stocks be measured as an abiotic resource depletion, which in LCA modelling is measured using an equivalence factor 'kg of Antimony equivalents'. This has the obvious disadvantage of trying to communicate water usage in terms far removed from the understanding of water users and the general public.

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. The Impacts caused by green water and non-evaporative blue water resources are not considered in the assessment.

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

Water footprinting and water scarcity – Ridoutt et al. & Pfister et al.

Ridoutt et al. (2009a, 2009b) have worked towards an integration of the water footprinting approach with LCA principles to enable determination of water use on water scarcity, with a demonstration of this approach for complex food products produced in Australia such as peanut M&M's and Dolmio pasta sauce. The authors state the value of such an assessment is in providing information to companies and the general public to identify water-use reduction opportunities and inform consumers of the environmental performance of products.

The authors have integrated standard water footprinting methods (described previously) with a LCA approach to system boundaries, functional units and treatment of co-products.

Their approach (presented in Ridoutt et al. 2009a, b) utilised a mix of actual water usage data throughout the supply chains of relevance, and literature data where real measurements were not available. Water usage is categorised as blue, green and dilution water, and these are measured separately.

As a hybrid approach, this study is not consistent throughout. For example, water used for irrigation is taken from real field application data for some crops, while for others a retrospective estimate is made from crop models of evapotranspiration. Moreover, the 'water use' estimates for irrigation that utilise real data represent 'irrigation volume applied to the field' with no consideration of whether the water was used by the plants for transpiration or was lost as seepage. Where irrigation water volumes were believed to have exceeded water requirements, green water was assumed to be zero.

While this approach may be suitable for broad scale assessments, it will not identify opportunities to improve water management at the farm level for two reasons, i) water loss pathways have not been quantified, and ii) green water has been excluded even though it plays an important role in the water balance and represents a possible source of renewable water for the agricultural system.

Ridoutt et al. (2009a) did not attempt to define the actual human or ecosystem health impacts of water use, and acknowledge the following limitations of the water footprinting approach at the product level:

- There is a lack of correspondence between water footprints and the availability of water for alternative uses in the absence of production,
- There is difficulty in relating water footprints to potential social and environmental harm.

These issues are particularly compounded by the addition of all water types (blue, green and dilution) into a single figure that represents the 'water use' of a product.

To extend this approach, Ridoutt et al. (2009b) identify that the main concern relating to water consumption in agri-food production is the potential to damage freshwater ecosystem health.

Therefore, they identify blue water abstractions and changes in blue water availability from land use change as the two related focus areas.

Ridoutt et al. (2009b) proposes removal of green water estimation from the calculus as unnecessary for the goals identified, with all attention to return to blue water. They consider green water to be ideally handled under the category of land use in LCA.

This approach appears very similar to Mila i Canals et al. (2008) described previously and has the associated strengths. However, Ridoutt et al. (2009b) proposes excluding green water from the inventory where Mila i Canals et al. (2008) do not. Ridoutt concedes the following limitations to their approach with relation to the exclusion of green water:

- It is not able to identify changes in water productivity in rain-fed production systems,
- It is not able to maximise calorific or nutritive value per unit of water consumed.

Considering the importance of green water in Australian agriculture and the key role it plays in the future of food production, it is a notable limitation to remove this from an assessment methodology. This is a case of moving towards an environmental impact assessment tool at the expense of a resource quantification tool.

A similar though more detailed approach has been presented in Pfister et al. (2009). This study again integrates virtual water measures with LCA, with the attention focussed solely on blue water use. However, they add to this a regionalised water stress measure, proposing 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.

Progress in this area of research is moving towards development of a 'stress weighted water volume' for a product (Ridoutt pers. comm.). This approach holds merit, as it will result in a single, comparable number that is understandable to the general public and has already taken into account the likely environmental impact of the water use. This approach is yet to be elaborated in the peer reviewed literature however.

Water balance modelling – Peters et al.

Peters et al. (2009b) have taken a distinctly different approach to assessment of water use, applying a methodology adapted to the Australian red meat industry context. This approach based calculation of water use at the inventory level on the results of water usage estimates provided by farmers, and through detailed, farm water balances. Some relevant water usage definitions used include:

1. Total water inputs, defined as rainfall, bore water, river irrigation water, reticulated (pipeline) water and water associated with purchases of livestock, feed or other inputs to the farm.
2. Total water extracted for use, defined as bore water, river irrigation water, and other reticulated water whether on farm or in upstream supply systems.
3. Total waste water generated, defined as animal urine and water with manure + effluent runoff at the feedlot.
4. Total water recycled, defined as the sum of evapotranspiration, evaporation (dams), respiration/perspiration, runoff and deep drainage.
5. Net water use estimate, defined as waste water and water retained in the red meat products exported.

The advantage of this approach comes in the high level of detail for the supply chain under study. Considering the majority of literature on water use in red meat production is based on calculation of the virtual water use, and that the majority of this water is derived from water used in growing forage and grain to feed animals, it appears appropriate to focus attention on these main sources of use. As these all occur on-farm (either the study farm or other farms used for forage or grain production) a farm water balance can be used to advantage. The water balance approach taken has been described previously in this report. Hence, the discussion here will focus on the strengths and weaknesses to this approach.

The water balance quantifies all flows of water within the system. This allows a greater sensitivity to possible impacts from water use or modification of the hydrology system. Table 35 provides a basic classification of the sources of water and the quality of the water after use in the systems is applied, reflecting the intent of key LCA authors interested in water use (de Haes et al. 1999; Owens 2002; Stewart & Weidema 2005). Surface water transfers of “flow” resources (de Haes et al. 1999), which contribute to reduced natural water flows in their original catchments, are grouped by a separate set of shaded cells. Water usage derived from funds was also identified.

TABLE 35: WATER USE CHARACTERISATION FOR THREE RED MEAT SUPPLY CHAINS (PETERS ET AL. 2009B)

(L/kg HSCW)		Victoria (farm + processing)		WA (farm + processing)		NSW (farm + feedlot + processing)	
		2002	2004	2002	2004	2002	2004
Water inputs source environmental characterisation							
Local catchment	Rainfall	7387	21541	57634	34922	17717	17684
Inter-catchment transfer "flows"	Agricultural irrigation supply	0	0	0	0	86	67
	Livestock or feed	0	0	0	0	233	217
	Reticulated supply	27	40	207	131	170	142
Inter-compartment transfer "funds"	Bore	0	0	7	4	51	37
Total inputs		7414	21581	57848	35058	18257	18147
Water outputs quality characterisation							
High quality	Evaporation and evapotranspiration	6664	14907	59171	33177	17219	16837
	Animal perspiration & exhalation	21	24	30	24	22	28
Moderate quality	Deep drainage	1122	6622	0	0	0	0
	Runoff	29	22	0	0	454	85
Low quality	Animal urination / excretion	42	49	18	14	30	45
	Discharge to sewer	3.67	3.30	3.35	3.33	3.33	3.33
Alienated	Water content of meat products	0.47	0.48	0.51	0.52	0.77	0.67
Total outputs		7881	21629	59222	33219	17729	16998
Total error	Absolute	-467	-47	-1374	1839	529	1149
	Relative	-6.3%	-0.2%	-2.4%	5.2%	2.9%	6.3%

Source: Reproduced from Peters et al. (2009b)

One obvious advantage to this approach is the ability to reclassify the results to compare with a broad range of assessment methodologies by handling the inputs and outputs in a different manner. For example, the blue water used for red meat production in the NSW supply chain, for 2002, is the sum of irrigation water use (on-farm), livestock or feed (use of commodities at the feedlot sourced from irrigated crops such as cotton), reticulated supply and bore water supply, equal to 540 L/kg HSCW, or about 3% of total water use. This is shown in Table 36.

TABLE 36: WATER USE IN RED MEAT PRODUCTION UNDER TWO DEFINITIONS (PETERS ET AL. 2009)

	Beef Production		Lamb Production		Beef Production*	
Production system	**Victoria**		**WA**		**NSW**	
Production year	2002	2004	2002	2004	2002	2004
Water use definition						
ABS - water transferred from source (Blue water)	27	40	214	136	540	464
“net use” - quality low or alienated	46	52	22	18	34	49

* Includes grazing and feedlot within the supply chain.

It needs to be clarified however, that total evapotranspiration figures presented in Table 35 represent the total evapotranspiration that occurred on the farm in the year of study. This is not comparable to the assumptions used by most virtual water studies, where green water is a measure of the modelled water requirement to produce the *required volume of feed to produce the animal under study only*. As described previously, this is done retrospectively based on crop models and is likely to result in lower estimates than presented in Peters et al. (2009b).

A second advantage is the opportunity to re-evaluate water usage data in the light of new considerations. For example, if attention is drawn to the groundwater recharge rates related to land use for red meat production, these data can be used to provide estimates of the quantity of water lost from the system via this pathway.

A detailed methodology of this type is also valuable for intensive production systems (feedlots) as it is able to determine the loss pathways, providing real solutions at the enterprise level for reducing water usage.

There are also some obvious limitations to this approach. Farm scale water balances, when generated for specific years, are subject to the climatic conditions experienced in that year. Likewise, when estimates are made on a functional unit basis, fluctuations in production from year to year may result in substantial changes to the results. The accuracy of the farm scale water balance is at the mercy of the model used to create the outputs; hence results must be viewed with caution. Additionally, it may be very difficult to extend this methodology to cover larger land areas, or to extrapolate performance across the industry.

The Peters et al. approach aggregated evaporative uses into the ‘recycled water’ category, based on the assertion that water evaporated or transpired is simply partaking in the natural water cycle and is not greatly affected by the production system under study. This is similar in approach to Ridoutt et al. (2009a) and Pfister et al. (2009) in one sense, though adequate differentiation between water sources (i.e. blue and green) was not undertaken. This also fails to take into account the role of LCA in quantifying water usage from a resource perspective.

Australian LCA methodology development

The Rural Industries Research and Development Corporation (with funding from MLA and others) have developed a methodology for agricultural LCA in Australia (Harris & Narayanaswamy 2009). The methodology is particularly focussed on greenhouse gas, energy and water use, and represents an adaptation of the ISO Standards to LCA in agriculture.

While providing some useful, broad scale guidance, the methodology fails to address the determination of green water within the inventory. The methodology does however specify the need for detailed estimation of on-farm water loss pathways such as evaporation, seepage and drainage.

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) 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; and
- 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:

- Water removed from rivers as a percentage of sustainable flow regimes; and
- Groundwater abstraction as a percentage of sustainable yields.

This may have some merit for its national relevance; however it does not follow the fairly established approach presented by Owens (2002) which has been used as a basis for most other water methodology developments in the field of LCA.

No studies are currently available that follow the proposed Harris & Narayanaswamy (2009) approach.

9.4 Comparison of Methodologies

Water definitions and assessment methodologies are created with a goal in mind for the application of the data. For the red meat industry, research to date (MLA projects COMP.094, FLOT.328 and B.FLT.0339) has aimed to provide the following outcomes:

- Assessment of the environmental impacts the industry is having on Australia's water bodies (generally defined as the impacts on water scarcity and ecosystem health),
- Provision of defensible water usage data for red meat production that can be used to inform the public, the industry and the government,
- Provision of an assessment framework that will be compatible with any government regulations in the future (of particular relevance to the feedlot sector).
- Identification of water usage inefficiencies within the production system and solutions to rectify these inefficiencies.

Additionally, research needs to be done in an efficient manner, and be representative of the wider industry for maximum application.

The projects completed to date have met these objectives to some extent; however the definitions and methodologies applied have not yet been subjected to critique in the literature. Broadly speaking, three weaknesses have become apparent in the projects to date, i) the comparability of the results to the literature. The majority of the literature for water use in red meat production is based on 'virtual water or water footprinting' and none of the studies completed adequately interpret the findings in this context, ii) the results do not adequately inform the industry and the general public of the environmental impacts from red meat production, and iii) the extensive study is not representative of the wider red meat industry.

Water usage definitions and methodologies have been presented from three broad perspectives, water engineering, virtual water and water footprinting and life cycle assessment. Of these, the water engineering methodologies for farm and catchment scale represents the traditional approach to water use assessment, focussed on liquid water sources or blue water. Alternatively, the virtual water concept has largely been developed to determine embedded water in traded agricultural commodities, which by definition includes water from all sources including rainwater. Life cycle assessment methodologies are comparatively less developed, and generally represent a hybrid of one or both of the other two approaches.

Based on the goals of previous MLA research and the broader context of Australian water policy and international research, comparisons can be made of the definitions and methodologies available for use in the red meat industry. It is unlikely that one single definition or methodology will be satisfactory at all levels, however it may be possible to present an over aching approach that is robust enough to meet a variety of goals as required.

9.4.1 Water Engineering and Virtual Water

As the two primary mechanisms used in the literature, the government and the general public, these will be compared with reference to the goals and needs of the red meat industry. The strengths and weaknesses of these approaches have been listed in Table 37.

TABLE 37: COMPARISON OF THE WATER ENGINEERING AND VIRTUAL WATER APPROACHES

	Water Engineering		Virtual Water and Water Footprinting	
	Strengths	**Weaknesses**	**Strengths**	**Weaknesses**
Methodology	Strong set of quantifiable definitions and methods within a limited scope.	General definitions and methods rarely present results on a functional unit basis and may or may not quantify rainwater in the system.	Able to determine 'real water' requirements for food production regardless of the origins of the water.	Lack of consensus regarding definitions and methodologies. Lack of specificity in water calculations.
Detail of assessment	This is the standard framework for irrigation management and water use efficiency. Able to identify water loss pathways and offer mitigation options at the farm level.	Site specific nature of the data collection and modelling may limit usefulness of the results at a broader scale unless this objective is specified – then catchment scale approaches may be applied.	Able to provide rapid global assessments that may be useful for water stressed economies and regions.	Retrospective methodology based on consumption requirements (forage and drinking) are not able to provide detailed results, or to identify inefficiencies in the production system.
Ability to communicate to general public and government	Engineered water 'use' is directly comparable to the understanding of water use by the general public and is the standard for government policy and initiatives.	Does not usually present findings for products, but rather for systems. This does little to inform the public of their water usage impacts from consumption.	Findings are catchy and achieve media attention – the water footprint concept is easy to grasp at the product level and will probably become well recognised.	Can be grossly misleading. Water 'use' is not comparable to the concept of water that the public have in their mind (which is blue water).
Water resource assessment	Able to determine accurately the use of blue water in a system.	May or may separately quantify the soil moisture (green water) from rainfall.	Quantifies the total water required in different countries regardless of source type. Rapid assessment possible.	Ambiguous when blue and green water are not presented separately.
Usefulness as an environmental indicator	Can be used at the farm or catchment level to determine flows that influence the environment, i.e. water abstractions, evaporative uses, seepage and deep drainage etc. Can be used at the catchment scale to balance needs between competing users.	Does not actually determine the environmental impacts, or likely impacts of water use at a specified level, i.e. it 'just presents the facts'.	Not the original intent of the tool. Can be used in water stressed economies to reduce pressure on local water supply through importation of water intensive products.	Lack of specificity and distinction between blue and green water will make results highly misleading for many agricultural systems, particularly in Australia where agriculture is highly reliant on rainfall rather than irrigation.

The virtual water and water footprint concepts were designed for a specific purpose that may not be closely aligned to the interests of the Australian red meat industry. However, in as much as the industry is a supplier of food to the world, the concept may be useful and of interest into the future. Moreover, if the concept becomes widely accepted in the public, the industry will need to ensure it is not misrepresented. Provided distinctions between blue and green water are made, and the environmental implications of these water uses are more effectively determined, the concept may be of use to the industry. Australian red meat may have lower blue water requirements than other regions of the world, maintaining Australia's 'green' image, now with respect to water usage. However, for most other goals the tool is not suitable.

The main uses of water engineering methods are likely to be at the farm level for detailed water accounting and identification of efficiency improvement options. At the catchment scale, there may be opportunities to calculate broad water use estimates for the industry by using water and livestock production data collected for national inventories such as the ABS. These will be subject to a range of inaccuracies inherent in the data collection processes; however it may be an interesting way to interrogate and extrapolate site-specific data to the industry level. These forms of assessment will be rigorous and can rely on the large volumes of data already collated by government agencies. The findings can also be easily communicated to the public and to the government to demonstrate the industry contribution to water extraction from water stressed catchments and aquifers.

9.4.2 Life cycle assessment

Though less developed than the other forms of assessment, LCA offers several advantages. LCA is a robust, systems based tool with a rigorous methodology for system assessment, boundary and functional unit definition, and handling of co-products. It has the twin goals of determining resource usage and the environmental impacts of a product or service. *Hence, it has as a primary goal the determination of environmental impacts, a major weakness for both of the other frameworks.*

LCA is generally a high level tool that allows for multiple approaches to data collection. This allows the tool to integrate other methodologies into the overall framework. This is seen in the literature, where LCA has borrowed methods and previously collected data from both the water engineering and virtual water or water footprinting approaches. This allows LCA to be adapted to the needs of the study in focus.

The main weakness of LCA is that a strong set of environmental impact categories have not been established yet. However, these are under development and will no doubt be established over the next several years. This will be a significant contribution to water assessment in agriculture. Provided the inventory phase of an LCA study is done with sufficient breadth and detail, multiple outcomes can be provided to meet the objectives of the study.

Considering water resource usage for example; by incorporating assessment of blue and green water, the study can present results that can be readily understood by the public and the government, and can be broadly compared with other industries (blue water usage). The study can also present findings of green water usage, allowing comparison with the virtual water literature. If detailed findings at the farm level are required, water balances or partial water balances can be used to identify inefficiencies. However, if these objectives are not understood by the researcher, they are likely to be missed. For example, application of a hybridised LCA / water balance approach may well simply estimate water usage for livestock drinking water by using standard 'text book' values. If no on-farm water balance data are collected, this may overlook major water use inefficiencies such as the use of uncapped open bore drains, where as little as 5% of the water extracted from the aquifer is used for livestock drinking requirements (Hassall and Associates 2003).

9.5 Preferred Water Usage Definitions and Methodology

Any methodology must be flexible enough to meet the multiple goals identified (and others identified by the industry) and practical enough to be used by researchers without the necessity for extensive research programs beyond the capacity of the industry to fund.

Life cycle assessment is considered the best over arching framework with which to study water use for the industry. The application of a preferred approach, using LCA is provided here.

LCA is moving towards agreement on basic parameters for assessing water inputs and flows, based largely on Owens (2002). This is the starting point for an assessment. In addition to quantifying water supply and uses in this way, data should be identified in such a way that it can be interpreted under the definitions provided by the ABS, which are essentially captured by the Owens approach. Water from blue and green sources should be determined to maintain maximum flexibility in the results. From this point, water data can be analysed using the impact categories proposed by either Mila i Canals et al. (2008) or Pfister et al. (2009). Further impact categories are likely to be developed in the near future also, and the most suitable selection can be made at the point of analysis provided the inventory has been appropriately developed and documented.

Data collection approach

LCA can easily integrate highly detailed site-specific data for foreground processes (i.e. on-farm production) with broad scale data for background processes (i.e. for products purchased onto the farm such as fertiliser). This allows the assessment to cover the whole life cycle without requiring excessive data collection.

Water use in the red meat industry will be dominated by the water used at the farm level. For this reason, it is recommended that further assessments of water use in the industry be based on a real assessment of water use on representative farms. This can be done through site appraisal, real data collection and creation of water balances that ensure major uses are not overlooked. Davis et al have developed and implemented a water monitoring framework in the feedlot sector. To improve the representativeness of these data, catchment water balance data may be used to capture water uses not found on the farms in question. Alternatively, these broader water balance or partial water balance data may be used to identify representative farms.

For the assessment of water use in processes that occur off-farm (such as the production of fertiliser for example) virtual water data may be utilised provided the quantity of blue and green water is specified. For many manufactured products the majority of water used will be blue water, and the overall contribution to the water use of an agricultural product is likely to be low. In many cases these data are already available. One exception to this is the handling of water use in feed production that occurs off farm. Because the contribution from this can be considerable, studies need to investigate water use for these products in Australia. This is a clear case where the livestock industries require investment in research by the grains and fodder industries to provide such data. Until this is developed, studies will rely on estimates made by water models such as those used in virtual water assessments. A simple improvement to this would be the estimation of blue water and green water components based on national data such as the ABS water accounts.

This approach is, in the authors mind, achievable for an LCA study. Detailed data for the feedlot sector have already been obtained, and a useful starting point has been established for the grazing sector. By supplementing actual measurements with broad scale data (such as ABS and catchment modelling) these estimates could be improved. Quality water usage data (green and blue water) are not available at the present time for grains and fodder however.

9.6 Reported Water Usage Estimates for Alternative Protein Sources

The comparability of alternative protein sources has been discussed elsewhere in this report (section 7.4). Several alternatives are commonly proposed to red meat, and these are discussed with respect to water use. In the absence of other data, virtual water has been used as the method for comparing products. Because of the noted variability in methods for calculating virtual water, comparisons are primarily made within studies that use the same methodology.

The weaknesses of virtual water have been detailed in the previous sections of this report and are noted where relevant in this section.

The most prolific source of data available with which to compare agricultural products has been compiled by Hoekstra and Chapagain (Hoekstra & Hung 2002, 2005, Hoekstra & Chapagain 2007, Chapagain & Hoekstra 2003b). Results from these authors are presented in Table 38.

TABLE 38: VIRTUAL WATER USE ESTIMATES FOR RED MEAT AND ALTERNATIVE PROTEIN SOURCES

Species	L / kg (Australian estimates)	L / kg (World average)	Reference
Beef	17,112	15,497	Hoekstra & Chapagain (2007)
Sheep meat	6,947	6,143	
Goat meat	3,839	4,043	
Pork	5,909	4,856	
Chicken meat	2,914	3,918	
Eggs	1,844	3,340	
Soybeans	2,106	1,789	

Table 38 shows the clear trend in virtual water use from very high (beef) to very low for eggs. As an alternative plant protein, soybeans are not significantly superior to the more efficient meat products, particularly if the protein content were taken into account. Australian beef production performed slightly poorer than the world average, while other animal products were superior. The reasons for this were not clarified in the source document however.

Pimentel et al. (2004) presents data for protein production in the USA (Table 39) which are significantly higher than those proposed by Hoekstra & Chapagain (2007). The reasons for these differences have been discussed previously in section 9.2.3. While the magnitude of these results is questionable, the trend is similar to the data presented in Table 38.

TABLE 39: ALTERNATIVE VIRTUAL WATER USE ESTIMATES FOR RED MEAT AND ALTERNATIVE PROTEIN SOURCES

Species	L / kg (USA)	Reference
Beef cattle	43,000	Pimentel et al. (2004)
Sheep	51,000	
Pigs	6,000	
Chicken meat	3,500	
Soybeans	2,000	

Key to any assessment of water use is a breakdown of water use by source between blue and green water. This is highly relevant to the comparison of water use from red meat as compared to other animal and plant products. Peters et al. (2009a) identified that the vast majority of "water used" in the production of red meat, according to Hoekstra & Chapagain (2007) was for forage and grain production. Water used for drinking (the only water that must be sourced from blue water reserves) will make up only a small fraction of the total water usage estimates however. It is not known what proportion of water used in the other livestock or plant protein sectors is derived from blue vs. green water. It is hypothesised that Australian products would show higher reliance on green water and lower reliance on blue water than many other regions of the world.

The implications on water use by changing protein sources away from red meat, as is often promoted in the media, are unclear. It may well be the case that this would save no water in the conventional sense (blue water) at all. In fact, it is possible that some irrigated crop products use higher volumes of blue water than red meat. This debate is not informed adequately by the virtual water or water footprint concept however, and until the impact categories under development in LCA are widely applied, the real consequences will not be known.

9.7 Conclusions, Knowledge Gaps and Recommendations

To develop a pathway to improve the efficiency of water resource usage and decrease environmental impacts the red meat industry must have a clear understanding of the strengths and limitations of the various methodologies used to present values for 'water use'. The various methodologies can be broadly grouped into three categories – water engineering, virtual water and water footprints and LCA. Whilst they have been developed for different purposes and may relate at some levels, they rarely relate at all levels.

The traditional approach to water use assessment adopted by private enterprises and governments is to define the quantity of water used in a particular locality (i.e. a farm, catchment, state), and 'water used' is typically the amount of captured, pumped or metered. Water balances are then applied to determine water use within the system at any scale, though the accuracy is dependent on the quality of the input data. Farm and catchment water balance estimates are typically made using models of hydrology and crop production.

The strength of this approach – when used for water accounting – is that it provides a full assessment of blue water movements attributable to a system, identifying where improvements can be made by reducing or eliminating losses. This approach has been used successfully in the feedlot and processing sectors to estimate water usage.

Based on the ABS definitions, water usage estimates for Australia's beef industry (from point source property data or a broad scale economic assessment) range from 27 to 540 L/kg HSCW. However, when first order estimates of the contribution of irrigation water to feed inputs for beef production (pastures and grains) were determined, the water usage estimate was 474 L / kg HSCW as a national average. This suggests water usage for beef production may be on the higher end of the range estimated by Peters et al. (2009a).

Many assessments of water usage for red meat production have been made using the virtual water and water footprint methodologies. These estimates vary greatly from 15,000 to 200,000 L/kg of beef and 6,000 to 51,000 L/kg of sheep meat. In most studies, the system boundary is unclear and blue and green water are included, however no distinction between them is made. Australian red meat LCA water usage data suggests that as little as 2% of the total virtual water used to produce beef in Australia is derived from blue water, with the balance being sourced from green water.

From a VW or water footprint perspective, meat is a more ‘water intensive’ product than a plant product and this has been used as an argument to reduce meat consumption. However, without knowing anything about the form of water used (blue or green), the land used in the production of the product (arable or non-arable) or other contributing factors, it is impossible to state that reducing consumption will result in genuine water savings.

Despite the obvious problems in interpreting these data for livestock production, differentiation of water type between blue and green sources is rarely presented in the literature, and is not well established in the available methodologies for calculating VW or water footprints. This is a flaw in the methodologies proposed, particularly for nations like Australia that rely heavily on rangeland beef production on non-arable land, where green water use is high and water has a low degree of transferability to use with other products.

A number of studies have identified the VW concept as an indicator of environmental impact, where the total impact on the environment is implied by the magnitude of the water footprint. However, few authors have been willing to elaborate on the environmental impacts of green water use for red meat production. This issue is integrally related to land use and land capability and a more accurate assessment will incorporate land use factors that differentiate land capability between arable and non-arable land.

As a trade tool for alleviating water stress by trading ‘embedded’ water with products, the virtual water concept has merit. However, as a proxy for the environmental impact that water usage has on aquatic environments (i.e. rivers), the concept is misleading when no differentiation of the source of water (blue or green) has been clearly elaborated and correctly interpreted in the results and discussion.

Therefore, the VW and water footprint concepts in their current form are not able to provide adequate detail to be of value in environmental assessments of water usage in red meat production in Australia. However, in as much as the industry is a supplier of food to the world, these concepts may be useful and of interest into the future. If these concepts become widely accepted in the public, the industry will need to ensure it is not misrepresented.

LCA is considered the best over arching framework with which to study water use for the industry. LCA is moving towards agreement on basic parameters for assessing water inputs and flows, based largely on Owens (2002).

Three weaknesses have become apparent in research projects carried out by MLA to date, i) the comparability of the results to the literature (the majority of the literature for water use in red meat production is based on ‘virtual water or water footprinting’ and none of the studies completed adequately interpret the findings in this context) ii) the results do not adequately inform the industry and the general public of the environmental impacts from water use in red meat production, and iii) the study that covered three grazing supply chains (Peters et al. 2009a) is not representative of the wider red meat industry.

Water usage definitions and methodologies have been presented from three broad perspectives, water engineering, virtual water and water footprinting and LCA. Of these, the water engineering methodologies for farm and catchment scale represents the traditional approach to water use assessment, focussed on liquid water sources or blue water. Alternatively, the virtual water concept has largely been developed to determine embedded water in traded agricultural commodities, which by definition includes water from all sources including rainwater. LCA methodologies are comparatively less developed, and generally represent a hybrid of one or both of the other two approaches.

A methodology has been proposed to improve the assessment of water usage in the red meat industry. The exact approach taken will depend on the future goals of the industry, however it now appears that the required methods are available in the field of LCA and through hybrid approaches using the methods of water engineering (at the farm and catchment scale) and virtual water (for determination of embedded water from inputs into the agricultural system).

9.7.1 Knowledge Gaps and Recommendations

The following knowledge gaps have been identified in this literature review:

- Detailed water use inventories for red meat that specify water by source and by type (blue and green water),
- Detailed water use inventories for major commodity inputs to red meat production such as grains and fodder (blue and green water),
- In depth review of broad scale Australian water use data (ABS and catchment scale water balances) to improve estimates of water use in the red meat industry.

It is recommended that, following review and acceptance of the approach presented, the industry conduct case studies to test its application within the beef and lamb industries. A first step would be to utilise the data already collected in various MLA projects and re-analyse these using ABS regional water use and production data for irrigated pastures and crops. Results could then be presented for blue and green water usage. Additionally, these data could be assessed using impact categories recently developed (i.e. Pfister et al. 2009; Mila I Canals et al. 2008).

10 Conclusions and Recommendations

10.1 Greenhouse Gas Emissions

Greenhouse gas emissions and carbon sequestration related to red meat production arise from a multiplicity of sources throughout the supply chain, with each being governed by specific conditions related to animal production, soils, manure and effluent, vegetation and fossil fuel energy usage. All these factors are influenced by variations in climate and management. As yet, many of the specific scientific research areas are still under development or are yet to be investigated under Australian conditions.

With such a broad scope, estimating emissions and defining research targets is a challenging task for the industry. This is further complicated by the range of estimation frameworks that are used by different sectors for their own purposes. Moreover, because GHG emissions are very difficult and costly to measure, most of these frameworks depend on estimation equations and emission factors that are inflexible with respect to alternative management practices and are not specific to the Australian climate or management conditions. Methodology frameworks also vary in the scope of emissions they cover. The Australian Government methodology for calculating contributions to the NGGI from red meat do not include emissions from fossil fuel consumption, soils or vegetation, while the NGERS includes only emissions from the burning of fossil fuels. However, a LCA on the other hand calculates all emissions (and can include sequestration) related to the production of red meat. The review of LCA literature showed that GHG emissions from red meat vary greatly across studies, from 8.4 – 28.7 kg CO_2-e / kg carcass weight (beef) and from 10.1 – 20.1 kg CO_2-e / kg carcass weight for lamb. The average emissions were 18.7 kg CO_2-e / kg carcass weight for beef and 15.2 CO_2-e / kg carcass weight for lamb. These results have been presented on an 'unallocated basis' *which is the least favourable comparative approach possible, with all of the environmental burden being assigned to the meat product at the point of slaughter.* None-the-less, it does offer a way to compare studies.

Considering the variability in methodology across these studies, comparisons between countries could not be made with any assurance. For studies that defined emission hotspots, the consensus was that enteric methane dominates overall GHG emissions, followed by nitrous oxide emissions. Following on from this it is clear that the breeding and finishing stages of production have the greatest impact on overall emissions.

Red meat production generally results in higher GHG emissions per kilogram of meat than other protein sources. Results from a literature review of meat LCAs identified chicken meat as the most carbon efficient meat (average of 4.2 CO_2-e / kg carcass weight), followed by pork (5.9 CO_2-e / kg carcass weight). Two studies that confirm this trend covered all four species with the same methodology, reducing the chance of errors that arise from methodology.

In comparisons with plant proteins most studies favour using a meal or complete diet as a functional unit to overcome problems with nutritional comparability. On a nutritional basis alone, plant based meals or diets are generally more carbon efficient, though they may use more energy during manufacturing. Studies that cover a wider range of environmental issues such as land and water use are not available however. Studies rarely take into consideration consumer preferences such as taste however, and the comparability of plant based versus animal based diets is highly questionable when a broader range of factors are included.

As with all expanding fields of research, the complexity and contrasting approaches to GHG estimation will diminish as knowledge grows. However, because of the pressures from the

Australian Government and the general public to estimate, report and possibly pay for emissions, the industry is seeking to find a 'clear way forward' for R&D in a short period of time.

Perhaps the clearest, overarching knowledge gaps related to GHG emissions identified by this review are:

- The need for accurate and understandable research results on emissions from livestock businesses and products, and
- The need for robust, flexible estimation techniques for key emission and sequestration sources from the red meat supply chain.

To address the broad scope need for GHG estimation throughout the industry, life cycle assessment is recommended as a research tool that has the ability to estimate all emission sources throughout the supply chain, providing reasonably robust results at the product (per kilogram of red meat) or business level if required. The data required for LCA may be used to provide assessments under alternative assessment frameworks such as the NGERS or proposed CPRS if required. LCA will also be able to quantify industry emission hotspots and show the overall potential of mitigation techniques to reduce overall emissions, allowing both industry members and researchers to understand where attention should be placed and the likely gains. This recommendation is supported by the technical reports on enteric methane emissions (report 2) and soil and manure emissions (report 3) which identify the difficulty in placing specific research in the broader industry context or comparing differing emission sources or mitigation techniques for effectiveness to overall emissions.

Numerous specific conclusions, knowledge gaps and recommendations have been identified in each chapter of this report and the supporting technical reviews completed for this project and are not repeated here.

10.2 Water Usage

Water usage in Australian agriculture is an important issue for the nation and has drawn considerable political and public attention. As with GHG emissions, calculation of water usage associated with red meat production is surprisingly complex and plagued by inaccurate data in the media and even within the peer reviewed literature. This review concludes that most inaccuracies relate to poor methodology or ambiguous definitions for water use leading to erroneous conclusions from research. There is on-going confusion between traditional water usage from surface or groundwater sources (so called 'blue water' use) and water usage figures that include rainfall to some degree (virtual water - VW). 'Green water' has been proposed as a new descriptor for soil evapotranspiration water derived from rainfall in order to help refine water usage estimates, though at this stage few VW studies differentiate between blue and green water. These form the extremes of the water usage methodologies and subsequent results, explaining why water usage for beef can vary from 27 L to over 200,000 L / kg HSCW.

Water footprinting is a third approach to water estimation that has been applied to Australian agriculture. This term has gained popularity in recent times and is being promoted by the CSIRO. Water footprinting methodology is still under development, with results that are generally lower than extreme virtual water estimates (17,112 L / kg Australian beef – carcass weight), most likely because of restrictions to the scope of water attributed to grazing livestock and the retrospective calculation technique. The CSIRO approach is working to combine water footprinting with LCA to indicate not only the water used, but also the likely environmental impacts of water usage in different regions.

Methodology is improving in this field and is at the point where hybrid methods could be applied to Australian case studies in the red meat supply chain to improve the quality of results available to the industry and public.

Life cycle assessment is recommended as the overarching framework to achieve this because of the dual focus on both resource usage and environmental impacts. LCA could be used to achieve a new approach that incorporates rigorous water balancing for the calculation of 'blue' water along with estimation of 'green' water requirements. Results can then be analysed to determine the likely impact of water usage on aquatic environments and to identify water hotspots within the supply chain.

Further, detailed conclusions and recommendations are provided in section 9.7.

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Appendix 1 – Policy contingencies for the livestock sector

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University of New England – July 2009.

The views expressed in this section apply to the authors only and do not in any way reflect the position of Meat and Livestock Australia (MLA) or the red meat industry.

Should agriculture be included as a covered sector in the CPRS?

This section outlines some of the policy contingencies which will determine the degree and distribution of impact on the livestock sector through Australia's emission trading scheme (the Carbon Pollution Reduction Scheme or CPRS). The focus is the legislated CPRS, with incidental consideration of the voluntary carbon market though the two forms of market will interact, particularly as the Australian government seeks to regularise the voluntary market, and as actions under the CPRS redefine the scope for voluntary markets.

Carbon pricing must inevitably impact on livestock production if the policy goals of significant reductions of carbon emissions to the air are to be achieved. The cost of emitting carbon pollutants will rise, significant adaptation and change will be required within the livestock production sector, and the additional costs will compound viability problems for many producers (Jiang et al. 2009). However, caution is justified in considering modelled impacts as innovation will alter the extent and incidence of costs. Technological options specific to reducing enteric emissions have been discussed above, and we touch on some other emission reduction or sequestration possibilities below. Further caution should be exercised where models do not take into account policy factors which may reduce the impacts, notably the announced intention to place a threshold on coverage by the CPRS. Modelling at this stage of the development of the CPRS is useful for highlighting issues, but a range of policy, technological and management contingencies that will alter that impact. This is generally acknowledged in reporting of models by their authors, but often ignored when discussing these results.

The IPCC Fourth Assessment Report (2007) took the view that despite significant technical potential for mitigation in agriculture, there has been little progress in the implementation of mitigation measures in agriculture on a global scale. Barriers to implementation are unlikely to be overcome without policy and economic incentives, regulations and other programmes, such as those that promote global sharing of innovative technologies (Section 10).

Many agricultural mitigation options also have co-benefits (improved efficiency, reduced cost, other environmental benefits) as well as trade-offs (e.g. increasing other forms of pollution), and balancing these effects will be necessary for successful implementation. Mitigation practices need to be evaluated for individual agricultural systems based on climate, soil type, topography, social setting, and historical patterns of land use and management.

The role of alternative mitigation strategies changes across the proposed range of price for carbon. At low prices, the dominant strategies would be those consistent with existing production, such as changes in livestock diet formulation. Higher prices allow for the use of more costly mitigation options.

The Prime Ministerial Task Group on Emissions Trading released its findings on 1 June 2007. The Task Group recommended that Emissions Intensive Trade Exposed (EITE) industries (with over a threshold of emissions) would be assisted to remain internationally competitive by being allocated free emission permits every five years, equivalent to the value of both their direct (changed industrial process) and indirect (flow-on impact of increases in fuel and energy prices) tax-effective emission abatement costs. New entrants to these industries would similarly be issued with free permits. These arrangements would persist for as long as other key nations do not impose comparable greenhouse emission constraints. Farms at present are not over the threshold and agriculture is not a covered sector.

Shortly after the Taskforce report, AFI (2007) produced a detailed report suggesting how agriculture could best act if it became part of a future emission trading scheme. The report suggested the activities that farmers could adopt at a farm level to non-permanently reduce greenhouse emissions was quite extensive and will evolve as research results progressively become available. Activities included adoption of minimum tillage technologies that reduce fuel use and soil carbon emissions, adoption of livestock management and feeding practices that reduce livestock methane emissions (covered in this report), changes to grazing and crop management systems that reduce soil carbon losses or increase soil carbon sequestration, adoption of fertiliser management systems that reduce emissions from that source and adoption of manure management or effluent treatment systems that reduce emissions.

Keogh (2009) reported that 3 separate modelling analyses have been carried out of the potential impacts of the (CPRS on Australian agriculture. These analyses were conducted by ABARE (Ford et. al. 2009), The Centre for International Economics (CIE 2008), and the Australian Farm Institute (Keogh and Thompson, 2008). The ABARE and CIE modelling use dynamic computerised general equilibrium models of the entire economy and different sub-sectors of agriculture, while the AFI modelling involves relatively static farm-level financial modelling.

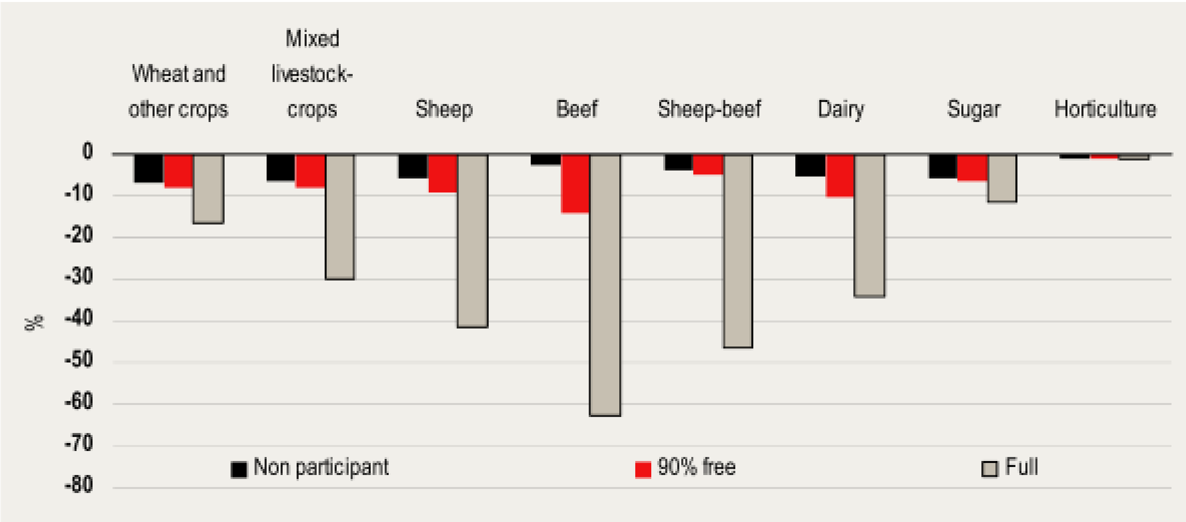

FIGURE 35: CHANGE IN FARM CASH INCOME UNDER DIFFERENT PARTICIPATION SCENARIOS (0, 90, 100% FREE PERMITS), $25/TCO2-E SOURCE: THE CIE (2008)

The results of these modelling analyses differed substantially. The ABARE analysis projected that the impact of the CPRS on agricultural production by 2020 would vary between +3% (grains) and -1.6% (other livestock), and by 2030 would vary between +5.3% (grains) and -8% (beef and sheep meat), relative to a business-as-usual scenario. Modelling by the CIE projected that the impact of the CPRS on agricultural production by 2020 would vary between approximately -1% (grains) and -9.1% (beef), and by 2030 would vary between -2% (grains) and -28.2% (beef) relative to a business as usual scenario. The AFI modelling found CPRS impacts at the individual farm level of up to an 18% reduction in farm cash margins by 2020 relative to a business-as-usual scenario, assuming a relative passive reaction to the CPRS by farm managers.

The majority of the differences arise from the assumption used in the ABARE modelling of equivalent international agricultural emission policies being implemented simultaneously with the Australian CPRS timetable in both developed and developing nations that compete with Australian agriculture in international markets. Other differences in results are likely to arise from assumptions in the ABARE modelling about the extent of development of carbon-sink forest plantations.

Based on this economic modelling Keogh (2009) argued that it would make no sense for the livestock sector to agree to participate as a covered sector in the CPRS after 2015, whatever the mechanism used to impose a cost on farm emissions. As a fully trade-exposed sector with limited mitigation options and as one of the least subsidised farm sectors globally, imposing even a partial emissions cost (in addition to the indirect CPRS costs that will arise in fuel, energy and processor pass-backs) on the sector would have drastic negative consequences for all livestock industries.

But, as is always the case, the issues are a bit more complex. The farm sector may not have a choice about paying for its emissions, as the Government has already indicated in Chapter 6 of its White Paper. Leaving agriculture's emissions out of the national abatement effort adds measurably to the cost of the Government achieving the fixed emissions target it signed up to in ratifying the Kyoto Protocol and the energy and mining sectors are already demanding farm emissions must be included. Agriculture's poor relationship with the majority urban population is also likely to be further weakened if the perception arises that the consumer's electricity and fuel costs are higher because farms are a big source of emissions and refuse to do anything to reduce them. The next request by the sector for drought support is also not likely to be favourably received if the perception is that farmers' inaction on emissions has actually increased the risk of drought.

Keogh (2009) stated that t**he issue is perhaps more accurately framed as - How, when, and under what arrangements could agriculture consider the possibility of a cost being imposed on farm emissions? It is unlikely that agricultural companies will be granted free permits to compete with their overseas competitors on a level playing field, as requested by R. Poole (Murray-Goulburn Dairy Cooperative) at the recent** AFI Agriculture, Greenhouse & Emissions Trading Conference, but it is possible that agriculture may negotiate specific arrangements for its inclusion in a trading scheme, due to its particular circumstances (I Carruthers, pers. comm.)

Conference participants identified a long list of pre-requisites that would need to be considered. These include comprehensive land system greenhouse accounting methodologies; realistic farm emission mitigation options; massive funding for R&D into both farm emissions mitigation and farm productivity (not substituting the former for the latter as at present); the development of workable systems to estimate and validate farm emissions; comprehensive economic modelling to look at the pros and cons of different CPRS-engagement models; realistic policy measures to prevent international leakage of agricultural emissions until the rest of the worlds farm sectors adopt similar policies; and an enormous and continuing communication and education program for farmers.

Can a work program involving all these elements feasibly be implemented? Who will take charge, and more importantly, where will the resources come from? Is it feasible to have all of agriculture (processors, financiers, farmers, inputs and service-providers) contributing to a program such as this and all heading towards an agreed objective? This was the critical question that attendees at the conference couldn't answer.

In the USA, agriculture and forestry offsets will be the oil that enables an emissions trading scheme to run smoothly (Miller 2009). The United States has not ratified the Kyoto Protocol, and as such emissions trading schemes established there are voluntary, though the emission reduction targets contracted by companies or individuals when entering the voluntary market are legally binding. In Australia, the proposed Carbon Pollution Reduction Scheme is Kyoto compliant, so is bound by the accounting rules associated with the Protocol.

The Iowa Farm Bureau established the first licensed aggregator of carbon credits on the Chicago Climate Exchange which is North America's only voluntary cap and trade scheme covering all six greenhouse gases. Today the Iowa Farm Bureau handles about 6 million carbon credits annually through its entity AgraGate. The United States has many regional initiatives already in operation such as the Regional Greenhouse Gas Initiative (RGGI), the West Coast Initiative (WCI) and the Midwest GHG Accord.

As the carbon market matures, more opportunities will emerge for agriculture and forestry (Miller 2009). Protocols for no-till, rangeland and afforestation management have been developed and implemented under the Chicago Climate Exchange, which has allowed the farm sector to learn by doing. Today over 9,000 landowners are involved across 35 states. Under the proposed cap and trade emissions trading scheme in the US there will be a threshold like that suggested under the Australian Carbon Pollution Reduction Scheme, which will result in less than 2 per cent of American farms being included (Miller 2009). Despite offsets reducing the overall cost of a trading scheme, shots have been fired at agriculture and forestry offsets because there's scepticism that they should be included and questions over whether there should be separate systems established for these offsets.

The fundamental principles

As this discussion will confirm, arrangements under the Kyoto Protocol, including the CPRS, are extraordinarily complex (Cacho et al. 2008, Aldy et al. 2009). However at the heart of the complexity of the Kyoto Protocol arrangements is a simple set of propositions, reflecting the belief that Adam Smith's "invisible hand" of prices will provide the incentive to bring supply and demand into alignment. In rural emissions trading, the demand that needs to be constrained is emission of particular greenhouse gases that are a by-product of farming; and the supply that needs to be increased is the removal of these gases from the atmosphere. The argument in favour of the use of markets in such circumstances is that they allow the least total cost reduction in environmental harm and that they are the best demonstrated way of stimulating private innovation to achieve this (Aldy et al. 2009).

The CPRS is only the latest in a series of markets for environmental services which impact on the farm sector, which rely upon a legally enforced cap on resource consumption or contamination. Tradeable entitlements to extract water, to emit salt, or (increasingly) to reduce biodiversity all use legal caps, trading and pricing to stimulate reduction of environmental harm. Necessarily there is a transfer of wealth between the seller of rights, and the producer and seller of credits, and an adjustment of competitiveness between those who can comply at low cost and those who cannot.

In properly understanding the impacts of the carbon market we suggest that farmers should not see this development in isolation. Some of the strategies that may be available to the farm sector are likely to involve considering bundles of environmental services together as either a buyer or seller of credits. Selling environmental credits may, in some cases, help offset the additional costs imposed by a CPRS (and the need to purchase credits for other environmental harms may increase the costs for other producers.

The cap on carbon emissions breaks down into a number of elements, represented by the following table of the behaviours that (from a carbon pollution perspective) ought to be either penalised or encouraged. There is a natural shift between the sides of this ledger, as action to reduce a cost is encouraged by the imposition of that cost (thus, avoidance of the penalty of the cost of burning fossil fuels, naturally encourages a switch to renewable energies).

TABLE 40: KYOTO INCENTIVES AND DISINCENTIVES

Purposefully penalised activities	Purposefully encouraged activities
Burning of fossil fuels (providing an incentive to reduce energy inputs and switch to renewables)	Increase permanent absorption of carbon in biomass and soil (an incentive to plant biomass that delivers long term sequestration and to reduce removal of such biomass)
Emissions of gas (methane) from animals (providing an incentive to reduce animal numbers and per-head emissions, and to capture these emissions)	
Emissions of gas (NOx) from land (providing an incentive to minimise nitrogenous fertilisers and to manage land to prevent emissions)	
Emissions of climate change gases from biomass rotting (providing an incentive to avoid reduction of biomass)	

Our discussion in this report has concentrated on issues specific to livestock production, enteric emissions. We have not dealt with the broader consideration of avoided fossil fuels, fertiliser use or the management of biomass to avoid emissions or to sequester carbon. Neither have we considered commercial responses to increased operating costs from carbon pricing which may offset the impacts of these costs.

Clearly, the higher the price of carbon, then the greater will be the incentive to reduce farm emissions and to increase farm sequestration, resulting in less fossil fuel burning, less methane from animals, less biomass rotting, more forestry and land management for sequestration, and greater permanence of carbon-embodied biomass.

An ideal system would achieve these goals at the least possible cost, in terms of farming activity, and the cost of operating and transacting within the market. It would also do so without imposing unduly upon any disadvantaged group in society (but without weakening the full incentive and disincentive effects of pricing of carbon). The nature of disadvantage is that those who suffer it are more likely to be unable to adjust to change circumstances, due to a lack of wealth or information, or because they are culturally constrained from rapid change (Martin et al. 2007).

Clearly, the more rapid the increase in the price of carbon, the greater the disruption of existing farm systems and, arguably, the greater the dislocation to those who are disadvantaged.

These observations highlight that the design of a carbon pollutant market involves fundamental political tensions in society: between those for whom a high carbon price represents opportunity and those for whom it is a threat. The larger the price of carbon, the greater will be the reduction of emissions. The faster that this occurs, the more rapid will be the adjustment. The larger the price and the faster the change to high prices, the greater will be the impact on all groups and these impacts are likely to be greater (proportionately) upon the most disadvantaged.

In absolute and relative terms (increased competitiveness of different farming enterprises or conditions) a CPRS will impact differently on different industries and people, and livestock producers are at an inherent disadvantage as their economic activity has higher emissions than many other farming activities. Even within the livestock sector, some will gain from having a restrictive system of carbon credits and debits, and others will lose because input prices will impact on competitiveness within the sector itself. It is, for these reasons, a CPRS is an intrinsically political and social beast, as well as an economically rational mechanism to pursue environmental sustainability.

Innovation and costs

Table 10.1 represents the intended pattern of incentives and disincentives to achieve a reduction of atmospheric carbon. As the discussion in the earlier chapters has highlighted, the livestock sectors, particularly cattle and sheep production, has higher carbon pollutant emissions than other farming due to enteric emissions. Without innovation and supportive policy arrangements, it would be consistent with the policy objectives of the CPRS for this sector to be impacted more heavily than other less emitting sectors. However, this preliminary conclusion discounts the opportunities for technical and market innovation, and the potential for policy arrangements to limit (or increase) these impacts.

The CPRS (and indeed the Kyoto mechanism itself) is at an early stage of its evolution. There is a well-documented pattern of evolution of innovations (including institutional innovations like new markets). In the early stages the focus is on invention and the refinement of the invention into being a practical solution. During this 'pre-paradigm' change, there is often an array of competing solutions being tried. We are at this point with carbon markets. Many early designs disappear as the more efficient ones arise, and the advocates of particular designs are strong in their claims that their model is indeed the future one. Gradually through trial and error and invention, a few designs emerge as the dominant paradigm. Further innovation refines these until they begin to approach the theoretical ideal. Ideas that were unthought-of of in the earliest stage of evolution come to be the dominant approach, and once 'core' technologies become a quaint symbol of past naivety. The evolution of the computer, with a large number of competing operating systems, 'clunky' and expensive hardware and storage media and high processing costs gradually evolving (with many failures but a great deal of innovation) into the types of systems we have today is illustrative.

The CPRS like many environmental markets is in this pre-paradigmatic stage (Martin et al. 2007). We see around the world great experimentation with environmental services markets of many types, such as pollution markets, fishing and hunting markets, water markets, and biodiversity markets. In the carbon market, alongside the formal Kyoto Protocol arrangements, there are a large number of voluntary offset schemes including the substantial private market operated through the Chicago Climate Exchange. Across eco-services markets there are many competing versions of the 'right' designs, a history of failures and successes, heated debates about design and confident predictions of how these markets will work and the impact they will have. The reality is that all assertions about the impact of the CPRS need to be understood as contingent on the dynamic of the process of innovation, testing, failure and refinement. They are also contingent upon the outcome of strongly contested political negotiations between countries and within Australia, which have yet to be resolved.

Kyoto (and post-Kyoto) policy uncertainties

Whilst the Kyoto Protocol is expected to be replaced in 2012 by an as yet un-negotiated international arrangement, the Protocol has set the parameters for international state-to-state carbon caps and trading arrangements that underpin statutory carbon emission markets. We will refer to this as the Legislated carbon market, to distinguish it from Voluntary carbon markets created by private forces[1].

In country to country negotiations there are many competing interests. One of the important issues for Australian farming is other country negotiators who see the need for a strong and restrictive cap to force up the price of emitting, or who maintain that any carbon credits for sequestration must satisfy a high burden of proof that credits will go only be awarded for highly secure, long term and measurable reductions in emissions (reflecting the stated intentions of the Protocol). They argue for restricting offsetting carbon credits, and point to past failings and uncertainties of forestry sequestration or industrial offsets for avoided emissions. Recently the concept of 'subprime' carbon has emerged to indicate the variability in the quality of claims for carbon emission offset credits (Friends of the Earth 2009). This contest of views over credits has significant implications for the Australian farm sector.

Broadly, for offsets to be counted for credits, the Kyoto Protocol requires that they be additional to what might happen if the Legislated carbon market did not exist, and that they provide technically and managerially reliable sequestration which is generally interpreted as requiring sequestration exceeding 100 years. There is a strong emphasis in the text upon scientific credibility and verification mechanisms. A concern for Australia will be the credibility of its claims for rural sequestration and avoided emissions. The scientific and institutional credibility of our proposals, which will also involve consideration of our performance to date. Important negotiation issues for Australian farmers[2] include:

- The rules for application of the "Australia clause" of the Kyoto Protocol (3.3), under which countries can claim credit for "verifiable changes in carbon stocks" due to avoided deforestation. Should it prove that deforestation has not been sufficiently avoided, or that verification is technically or administratively unreliable, then renewed pressure to limit this offset and to tighten its rules should be expected. If this were to occur Australia's capacity to comply would fall markedly, with consequent substantial impacts on all industry. This suggests that avoided land-clearing will be increasingly important to industry and government.
- The rules for recognition of forest carbon sequestration, where there are complex science issues about the rate and permanence of sequestration for different forest types and management regimes. There are also proposals by the forest industries for recognition of construction timber as a sequestration method. Regardless of the outcome of such developments, tighter rules for forestry sequestration and accounting for carbon sequestered should be anticipated. New techniques (for example remote sensing and improved statistical estimation) may however allow a reduction of the transaction costs of such accounting.
- The expansion of land management sequestration options, with particular emphasis on the rules for recognition of non-forestry biomass, soil biomass and the extension of credit arrangements to recognise 'bio-char'. Abuses of the carbon credit rules in the international market have created a suspicion of further liberalisation of avoided emission and

[1] Noting that with action to ensure the credibility of Voluntary markets and to extent Legislated markets, these distinctions are likely to diminish over time.

[2] For a detailed examination of these issues, see Cacho et al. 2008.

sequestration credits (Friends of the Earth 2009). The scientific credentials of the extent and duration of sequestration or avoidance, security of risk-management, and strong institutional arrangements are likely to be essential to any extension of farm-based sequestration or emissions avoidance credits.

- Given indications of adverse social effects in developing countries (in particular) from carbon-reduction strategies such as plantation forestry and biofuels, arrangements to strengthen protection for environmental and social values should be anticipated. Reducing the environmental risk of monoculture carbon plantations, or impacts of bio-energy strategies (such as biofuels subsidies and mandates) on the environment and on the poor, may complicate farm-based sequestration or emissions avoidance.
- Clarification of the extent to which a country or an industry (or a firm) may use purchased credits to offset their emissions. The Kyoto rules suggest somewhat vaguely that purchased credits ought to be supplemental to actual reductions and sequestration by the emitting sector. Clarification of the rules may limit the ability to use purchased credits from the farm sector.

Restrictive rules will limit the offsetting opportunities for the Australian farm sector, and lead to a higher price for carbon emissions. However, those who do produce recognised carbon credits are likely to enjoy higher prices.

International negotiations will also affect the transaction costs for approved emission avoidance and sequestration credits. Mandated methods for identifying, measuring, reporting and trading under the Protocol will determine the cost and complexity of emissions accounting and trading. If (for example) soil sequestration or sequestration through bio-char were authorised for credits subject to detailed site-specific measurement and risk-management, then these options will be less viable for farmers. Similarly, if carbon credits from forestry required stronger protection against de-sequestration (for example in the event of fire) then more robust insurance and other risk arrangements would be required, reducing the economic value of the credits. A balance will have to be struck between the desire for precision and security of emissions avoidance and sequestration; and the desire for low transaction costs in securing these credits. Where this balance lies will be contested in forthcoming international negotiations.

Design of the Australian market

Countries are free to design their own systems to achieve their Kyoto protocol commitments. Both sides of parliament support an Australian carbon emissions/sequestration credit market, and the legislation for the preferred model of the government has been tabled. However, there are many uncertainties. There will be adjustments and compromises during the political processes of passing the laws and further amendment thereafter, notably with any change of government. The announced intention to defer decisions how the farm sector will be included until 2013 (for introduction in 2015), leaves space for many key aspects to remain uncertain for some time[3]. Some key issues are outlined below.

[3] Though the degrees of freedom surrounding this choice will be increasingly constrained, as firm positions are locked into both the international and Australian carbon market regimes.

TABLE 41: KEY ISSUES RELATING TO THE DESIGN OF A CARBON TRADING SCHEME FOR THE AUSTRALIAN MARKET

Design element	Possible implications
International trade in credits.	A national system that allows overseas credits to be counted by local emitters would probably reduce the value of credits. This will reduce the total impact of carbon costs across the economy. It may reduce the value of on-farm credits for avoided emissions or carbon sequestration.
Thresholds for emitters	The transaction costs and complexity of having every emitter account for their emissions would be massive. For this reason, it is proposed to have a 'threshold' below which emitters will not be required to individually account. The threshold proposed for Australia is 25,000 tonnes of carbon dioxide equivalents (CO^2-e). Adjusting this threshold will affect who is required to account, and the cost and complexity of the scheme.
Tax or trade approach	An alternative to a carbon market for the farm sector is a carbon taxation system, possibly with the tax income being 'hypothecated' to support emissions reduction and sequestration and/or adjustment by the farm sector. These choices will impact on administrative cost and on the net impacts on the farm sector (and on the speed of adaptation).
Permit property and accounting	Whilst states have passed laws to confirm their overarching ownership of carbon in trees (mirroring public ownership of water, whilst allowing for trade in rights to extract water); the system of recognition and recording of rural carbon debits and credits remains undefined. This will impact on complexity and transaction costs, and on the extent of economic costs and opportunities for particular enterprises.
Incidence, payment and reporting	There are many different possible points at which carbon credits may be levied and accounted for. They may be levied upon inputs (e.g. fertiliser and fuel), or on estimated emissions (for example based on per-acre or per-head of livestock), or on some other basis.
Recognition of farm carbon credits	Whilst the international rules for recognition of credits and debits will be binding on countries, it is possible for a country to impose its own mechanisms for recognition. The rules for counting emissions, avoided emissions and various forms of sequestration will impact on the availability of carbon credit income for avoided emissions and different forms of sequestration.
Risk arrangements	In principle, de-sequestration (for example through a fire or through harvesting) should result in the need to refund any credit payment achieved or purchase credits. However, the potential for these issues to compound natural disasters and increase uncertainty for the farm sector is likely to lead to innovative risk arrangements being developed.
Implementation within natural resource management	Whilst not strictly a CPRS design issue, the extent to which institutional arrangements for various environmental service markets (e.g. water, carbon, salt, biodiversity) and regulations are integrated will impact on the economics and complexities of farm based natural resource management.
Biofuel mandates and subsidies	Avoided burning of non-renewable carbon fuels represents a reduction of emissions. This fact, plus a number of other considerations such as fuel security and rural industry support has created an impetus for biofuel mandates and subsidies. These interventions, coupled with the rules for recognition of biofuels for carbon credits, will determine the attractiveness of this mechanism for carbon emission reduction.

These are but some of the government-level institutional design elements that have the potential to radically alter the incidence, and net cost, of emissions markets to the farm sector.

The detailed design challenge

For the CPRS, the 'devil will be in the details' in terms of its impact on the farm sector. Prior experience with water and other eco-service markets illustrates that four 'architectures' have all to be effective for a market to meet its goals. These are

1. The technical architecture, principally concerned with the concept for the market.
2. The institutional architecture, principally concerned with the design of trading, measurement and governance structures.
3. The legal architecture, principally concerned with the design of the statutes, regulations, contracts, conflict resolution arrangements and detailed rules and processes for implementation.
4. The administrative architecture is principally concerned with the detailed implementation of all of the above.

The technical architecture is the most conceptually open and intellectually engaging for most people, whereas the lower rungs of this ladder are the most detailed and painstaking. To date most of the work on the CPRS has been focused upon the 'big picture' issues of Australia's emissions avoidance and carbon sequestration strategies, such as the relative role of trading versus tax or regulation. However, the legal and administrative architectures are likely to have the greatest impact on the complexity and transaction costs for those who seek to trade in the market. *Ex-ante* modelling of the technical architecture may indicate a theoretical potential for gains from trade, but *ex-post* it is the legal and administrative arrangements that often determine whether the market will work in practice.

Whilst there will continue to be a necessary debate about the technical and institutional architectures, regardless of how the details are resolved some elements will almost certainly be essential. Detailed issues will include: refinement of property rights; a range of contractual and trading mechanisms; potentially licensing of traders or other market participants; registration of interests; electronic trading arrangements; audit structures for various forms of carbon credit; and financing arrangements for these market arrangements.

If these elements are not in place ahead of trading commencement, it is possible to anticipate:

1. Dissatisfaction and loss of confidence. Implementation questions will become more pressing and contentious the closer that Australia moves towards a launch.
2. High transaction costs, particularly in the early stage. This can sufficiently undermine trader confidence and tax the value of transactions to such a degree that it prevents trades occurring.
3. Stress and conflict, and a messy process of 'catch-up' which can in turn result in permanent implementation problems as compromises are made in an attempt to patch design failings.
4. Excessive stress on the institutions (and staff) of government whose task it is to ensure effective implementation of the technical and institutional designs.

Whilst there will always be some degree of confusion, complexity and conflict with the creation of a new market, it is in the national interest to minimise this by careful design, and by building implementation capacity well ahead of it being needed. The discussion which follows considers some of these detail issues.

The point of obligation debate [4]

Illustrative of the impact of detail on the costs and effectiveness of the CPRS in relation to agriculture is the question of the point of obligation for reporting and ETS credits or debits to be accounted. Liability could be imposed:

1. directly on farm businesses;
2. indirectly, on ‘up-stream’ inputs such as fertiliser and/or on ‘down-stream’ food processors such as abattoirs; or
3. via a hybrid of these approaches whereby the default point of liability would be up or down stream but farm businesses are given the option of managing their emissions liabilities directly.

The bulk of the sector’s emissions are produced by thousands of small farm businesses potentially making it costly and inefficient to impose obligation at the farm level. Few farm businesses would meet the proposed minimum 25kt CO^2-e threshold under the National Greenhouse and Energy Reporting Scheme. This, coupled with issues of transaction costs and, suggests the potential inefficiency of imposition at the individual farm level.

Counter to this is the argument that management action and farm specific characteristics significantly affect emissions associated with production and input variables such as meat or milk production or fertilizer consumption. The scheme will be more equitable and efficient where emissions can be estimated accurately and cost effectively at the point of emissions because: the emissions liability matches the actual emissions at the site; and as a result there are more opportunities for individual entities to respond to the carbon price by changing their behaviour or technology.

However the transaction costs of the system are strongly influenced by the number and scale of the reporting entities. There are around 130,000 enterprises in the land-based sector. These vary in scale from ‘hobby’ farms to large corporate operations. Table 10.2 shows indicative emissions by industry sub-sector and the approximate number of entities responsible. If an emissions threshold were to be used for the agriculture sector, thresholds would need to be set at a relatively low level to capture the majority of agricultural emissions. For example, covering about 80 per cent of direct emissions from the beef, sheep, dairy and wheat industries would require participation of around 45,000 farm businesses.

Use of thresholds introduces the possibility of economic distortions between entities above and below the relevant threshold, because carbon costs are not imposed on below threshold entities. This could create incentives to change company structures.

[4] This issue is discussed in detail in Agriculture Technical Advisory Group (2009)

TABLE 42: AGRICULTURE SECTOR PROFILE (MAJOR INDUSTRIES) SOURCE: DEPARTMENT OF CLIMATE CHANGE (2006D)

Sub-sector	Number of entities	Annual emissions (2005) Mt CO2-e
Beef cattle	36,000	42
Dairy cattle	9,900	10
Wool	13,000	17
Sheepmeat		
Mixed sheep/beef cattle	8,300	Included in other sub-sectors
Mixed grain, sheep/beef cattle	17,200	Included in other sub-sectors
Pigs	900	1
Poultry (meat and eggs)	1,100	1
Grains	12,700	4
Sugar	4,600	1
Cotton	600	<1
Rice	2500	<1
Horticulture & fruit	20,000	1
Total	**130,000**	**79**

To achieve comprehensive coverage of all emitters (both large and small) where the costs of coverage are excessively high, an option is to cover emission sources indirectly. This could be achieved by requiring up or down stream entities, such as fertiliser distributors or food processors, to acquit scheme units for emissions from consumption of their products (upstream entities) or production of their inputs (downstream entities), using proxies of direct end use emissions. This is the approach that New Zealand has proposed for its emissions trading scheme.

Provided that the link between the upstream activity and emissions is unbiased, from an economic perspective, incentives to reduce sectoral emissions will be present. However, as noted, these are likely to be more muted under an upstream approach. There is potential, though, to provide greater mitigation incentives through development of more differentiated emissions factors. Not all emissions sources could be captured via up stream and down-stream activities. For example, breeding animals and animals slaughtered for on-farm consumption would not enter the supply chain and so could not be covered 'downstream'.

The costs of an indirect approach will depend on the number of liable entities, which in turn will depend on the number of covered emissions sources and the precise point of obligation. Overall costs would likely be significantly lower because there would be hundreds rather than thousands of liable entities. Development of an indirect approach to coverage would require identification of points of obligation that efficiently and comprehensively cover emissions from the sector.

Indirect liability would avoid the problem of economic distortion between farms on either side of the emissions threshold because the up (or down) stream entities would supply to (or receive produce from) farms of all sizes. It may be difficult to identify practical indirect points of liability for all emissions sources. For example, breeding animals and animals slaughtered for on-farm consumption would not enter the supply chain and so could not be covered 'downstream'.

A further option is for liability to be imposed indirectly ('upstream' and 'downstream') as a default option, but farm businesses given the option of reporting and managing their own emissions liabilities (accepting direct liability). This option could be a way of obtaining the advantages of both direct and indirect approaches.

Alberta is the first province in Canada to legislate GHG reductions. Starting on July 1, 2007 all companies that emit more than 100,000 tonnes of GHG a year must reduce their emission intensity by 12% per year starting in 2007 (Alberta Environment, news release, March 8, 2007; Bill 3, Climate Change and Emissions Management Amendment Act, Specified Gas Emitters Regulation). *Companies can reduce their emissions intensity by making operational improvements, by buying an Alberta-based offset to apply against their emissions total or by contributing to a government fund that will invest in technology to reduce GHG emissions in the province*. Buyers and sellers of GHG credits can register on-line with Climate Change Central, Emissions Offset and Trading (http://environment.alberta.ca/1238.html). Sellers must follow one of 24 quantification protocols which are filed on-line at the above mentioned website. There are currently three registered protocols relating to beef cattle.

The impact of private entrepreneurship

The discussion above has focused on the potential impact of public institutional arrangements on the cost, complexity and impacts of the future CPRS. However, to fully appreciate the possibilities it is necessary to consider these institutional design issues within the context of potential private responses to the costs and opportunities that a CPRS will create[5].

Perhaps the most significant reason for using markets rather than regulation and taxes is the entrepreneurial energy and innovating capacity of the private sector. The role of government is principally to set credible and strong institutional structures within which the private sector can invest, innovate and trade.

The carbon emissions avoidance/sequestration market in Australia has largely grown outside the Kyoto protocol and arose ahead of the proposed CPRS. It exists in three forms – state required emissions offset for state owned electricity generators; international purchases of sequestration within Australia that are recognised in the country of the purchaser, and the voluntary carbon reduction market. Internationally, there is a fourth non-Kyoto market, the Chicago Climate Exchange (CCX). This market is extensively involved in farm sequestration and emissions offsets and interestingly offers a significant program of carbon credits for soil sequestration.

What is of greatest interest here is the Australian voluntary carbon market, which is made up of some 50 distinct operations each of which has its own methods of operation (and different prices for carbon sequestered). The advent of a national Legislated emissions reduction and sequestration must impact on the Voluntary market, but it is far from given that the voluntary market will disappear. The history of innovation would tend to suggest that whilst many weaker early entrants may not survive as independent operators, a small number will innovate and ‘morph’ to dominate segments not adequately covered by the Legislated carbon market, or will innovate and adapt to become successful within the legislated framework. The evidence is also clear that faced with significant economic incentives (positive or negative), including those generated by environmental regulation, inventiveness will be stimulated and un-anticipated inventions and innovations will emerge.

Research and development to address key challenges for the farm sector is underway in Australia and internationally. The foci include:

[5] For a detailed discussion of the effects of institutional incentives on innovation in Australia see Marceau et al. (1997).

1. technologies to reduce on-farm energy needs, and to shift to non-fossil fuel sources. Coupled with work on cleaner electricity production, research and development are underway addressing equipment fuel consumption, and on-farm energy production (including second generation biofuels and pyrolysis).
2. methods for addressing enteric emissions from livestock, including potential inoculants and dietary additives, genetics, and emissions capture in intensive production systems.
3. production and business models for forestry sequestration, including new 'engineered' on-farm plantations, improved species and more efficient methods of assessment of carbon sequestration.
4. development of the science upon which to advance and evaluate Kyoto-compliant carbon credits for long term non-wood biomass and soil sequestration, biochar, and for construction timber to be included as a sequestering medium.
5. Methods for lower cost and reliable metrics for emissions reduction and sequestration.
6. Innovative commercial structures, including insurance products against de-sequestration, new derivatives (options, futures and revolving shorter term credits), investment structures such as multi-attribute markets which integrate carbon trading with water, salt and biodiversity values, and integration of the Legislated market with the Voluntary market with its' less constrained capacity to create new credits and trading arrangements.

The results of these innovations will inevitably compound. New technology solutions will be married to new trading and investment structures, and new farming business models will emerge that integrate emissions reduction and sequestration as part of the farm business model. There are many pioneering developments underway already, that indicate this potential.

The impact of the CPRS on livestock producers?

The impact of the CPRS will be upon not only the economics, but also the operations, of livestock producers. The CPRS is intended to, and will, increase particular costs of livestock production as a means for motivating farmers to reduce emissions and increase on-farm sequestration. Regardless of the way in which the legal and institutional issues we have noted are resolved, carbon emissions controls must lead to:

1. increases in the cost of fossil fuel energy, either petroleum products or electricity.
2. greater economic and managerial pressure to reduce or capture enteric emissions from livestock.
3. increasing costs of nitrogenous fertiliser and greater management pressure on the efficient use of fertiliser to minimise emissions.
4. increasing incentives and pressure to reduce land clearing and to increase on-farm vegetation.

These four imperatives will result in marked changes in farm systems. Overlaid on these it is likely that farmers will see

1. intersection of emission and sequestration trading or taxing arrangements with natural resource management approaches for water markets and biodiversity conservation (both market and regulatory) and salinity and drought arrangements.
2. a proliferation of technologies and farm management methods to address these challenges;
3. new trading and investment products and services providers; and
4. government support schemes for the transition.

It can also be expected that responding to these new pressures will impact on the complexity of enterprise management, and require further capital investment in new infrastructures, and in purchase of environmental services credits. It can be expected that the spread of costs and opportunities will be highly varied due to different farm conditions, enterprise types and farmer capacity.

Increasing carbon-resilience of the livestock sector

The impacts of carbon emissions and sequestration trading under the Kyoto Protocol will not be distributed evenly across nations, across industries within nations, within the farm sector or even across farms within sub-sectors such as livestock production. Some causes of variability are outside of the control of producers or the industry as a whole, but there are many ways in which livestock industries and individual operators can reduce risks and improve potential opportunities. What is needed is a multi-level strategy to provide the maximum potential for the farm sector to fully respond to the incentives and disincentives generated by emissions control mechanisms, with the least possible inequity in the distribution of costs and benefits of both markets and regulatory controls.

The following table is indicative of the type of multi-level strategies that will be required if the sector is to best equip itself to handle the many contingencies which will ultimately determine the impact of the CPRS and related developments on the sector.

Level of action	Proposals for livestock sector action
International	To maximise the opportunity for innovation in farm-based credits for sequestration, avoided de-sequestration, and avoided emissions Australia should seek clear, credible and flexible principles for the creation of new Kyoto approved credits, and seek to have the role of this approval delegated to an expert group.
Trans-national	To avoid the possibility of unfair competition arising from different intra-national emissions and sequestration management frameworks, Australia should review its' customs anti-dumping arrangements. Because the extent of fungibility of Australian sequestration and emissions control carbon credits with international credits will impact on the cost of farm emissions and the value of farm credits, careful modelling of the impact of recognition of international credits should be conducted before this policy is set.
National and industry	Neither the farm sector nor the community will benefit from a further proliferation of fragmented mechanisms that seek to improve the public good - carbon credits and management arrangements are intrinsically inter-related with other on-farm environmental services in terms of outcomes and impacts on the enterprise. - Because of the importance of reduced fossil fuel emissions and links to the farm carbon budget, biofuel mandates and/or subsidies, and fuel tax arrangements, and regulations to prevent perverse effects, must be integrated into the design of the rural emissions control and sequestration enhancement strategy for the rural sector. - Complexity and transaction costs, and reduced value of outcomes, arises from the fragmentation of regulation and market instruments

	for on-farm environmental outcomes; and reduce the potential for farmers to benefit from good environmental management. - Property rights, registration and measurement, trading arrangements and regulatory oversight of farm-based environmental services should be integrated and streamlined, to maximise opportunities for farmer participation and innovation. The high transaction costs of registration and monitoring of carbon credits (as with other credits) erode the economic value to farmers from their public good contributions. More economic metrics and data, better methods, and innovations to achieve economies of scale will require private innovation and strong support by government. A coordinated strategy to reduce farm carbon transaction costs is needed. Relative to other foods, sheep and cattle meat contains a high level of embedded energy and water and this will reduce its relative competitiveness as a food. However, with increasing wealth around the world the potential remains for the industry to sustain growth and profits provided that it can maintain its market presence and improve production efficiency. An important part of the adaption strategy for the industry must be continued emphasis on farm management, production efficiency and strong marketing. The push to 'de-carbon' farming will increase emphasis on the low-cost and carbon-saving transport and other systems. Opportunities for using carbon issues to elevate rail and other transport efficiencies (and to secure carbon offset support for these investments) should be developed. Taxation arrangements need to be realigned to support the transition to a different relationship between farming and conservation (including carbon emissions reduction and sequestration). Improvements ought encompass reconsideration of the definition of forestry (to encourage non-harvest forestry), treatment of environmental philanthropy,
State	If the farm sector is to rapidly transform itself to meet environmental public good expectations, it must obtain the economic benefit of its efforts, notably: - that the economic and environmental benefits to industry and the community from avoided land clearing and biodiversity conservation on farms be reflected in payments to the farm sector. - that market instruments for biodiversity conservation such as bio-banking, combined with sequestration and avoidance credits, be targeted to maximise the potential for farm-based eco-service delivery.
Region	Regional opportunities - shared facilities e.g. biofuel, pyrolysis, education, access to government investment - re-tasking of regional natural resource management agencies and investment priorities, to accommodate the increasing centrality of economic interests and private investment in both conservation and farming

<table>
<tr><td>Landscape</td><td>There is a greater potential to maximise the gains and minimise the risks of environmental services, including avoided emissions and sequestration, by management on a landscape scale across multiple (public and private) land tenures. This creates possibilities for cost sharing, connectivity, and risk-management such as fire control. It also provides the potential for greater transaction economies in participating in carbon and other environmental services markets. To achieve this will require that landowners (possibly working with local government and regional agencies, and public land managers) develop networks and coordinated strategies to maximise the economic opportunities from landscape-scale delivery of environmental public goods.</td></tr>
<tr><td>Enterprise</td><td>At the enterprise level, there is much that can be done to prepare for the new situation. The enterprises which are likely to be best situated for the longer term will
1. be naturally (or managerially) endowed with good soil and water, infrastructures and access to markets;
2. have high quality management, who are aware of the issues and prepared to be positive in the pursuit of opportunities;
3. be in touch with government and private sector buyers of farm environmental services, and providers of transition support like grants, networking, and knowledge;
4. have highly efficient energy use, which will involve suitable equipment and energy management systems;
5. use minimal nitrogenous fertiliser, and manage that fertiliser to reduce ‘outgassing’ through pasture and stock management;
6. have farm-forestry and other environmental services (such as biodiversity or salinity controls) embodied in their farm management programs;
7. ensure that their environmental credits and opportunities for credits are well documented and registered, so that economic opportunities are not lost.</td></tr>
</table>

Carbon pricing is intended to increase the costs of activities which emit such as vegetation removal, use of fossil energy, and the use of fertiliser. It is also intended to provide an economic incentive for activities that will prevent loss of vegetation and which remove carbon and prevent emissions to the atmosphere. Responding to these requirements will also impact on capital structures and enterprise management. Livestock production is intrinsically at a disadvantage compared to many other farming activities, but that is not to say that this disadvantage will be excessive if the industry and managers of enterprises rapidly move to a prepared footing.

A passive or defensive stance at this time of great change is likely to be a recipe to ensure that one’s fears are realised. A positive approach to shape that future, and to ensure that the industry and the enterprise are well placed to be resilient, is the most significant requirement for the livestock sector at this time.

Summary and Knowledge Gaps

Livestock production is emissions intensive, and therefore it will be impacted directly and indirectly by any carbon-pricing mechanism. This will impact on competitiveness with other food and fibre sectors, and it will also impact on regional and international competitiveness. The net competitive and economic effect will depend upon these factors and the ability of the sector to maintain or improve its relative position by ‘traditional’ strategies for marketing and efficient production.

Whilst models of carbon market impacts on the farm sector are a necessary part of developing rural carbon policy, all are based on unavoidably unreliable assumptions about the legal and institutional settings. Carbon market policy settings are being (and will continue to be) hotly contested from many powerful competing interests and the outcomes for the farming sector on key issues like rural sequestration are indeterminate.

Key variables include:

- international fungibility (ease of exchanging one unit with another unit of the same commodity) of credits and debits,
- thresholds and points of incidence,
- the use of taxes or trading,
- permit property rights and trading systems,
- the recognition of farming-offsets such as biochar or soil carbon,
- de-sequestration or market failure risks,
- policy constraints on the operation of the market, and
- issues impacting on transaction costs.

The dynamics of the Kyoto-based market will also impact on the viability of voluntary markets, and it should be expected that there will be significant competitive change (and possible ‘shakeouts’) in the non-Kyoto sector.

There will be viable technological and managerial innovations, many of which are already being developed or trialled, likely to moderate and shift the incidence of costs and benefits. These include enteric-emissions control, but also livestock emissions capture and reuse (by the use of sheds and emissions capture), farm-based biofuels, engineered woodlands; and also the integration of environmental services with the control of carbon cost. The ability of livestock sectors, or producers, to adapt will be a key to the actual impacts. As a result the distribution of costs and benefits are likely to be unequal.

This section outlines elements which could be considered in the development of an integrated and proactive response by the livestock sector. These span international negotiations, national and state issues, and preparedness actions at the farm level. It is clear that focusing on the Australian farm-sector CPRS is a necessary, but far from sufficient response, to the uncertainties.

This section of the report is packed with unknowns, but many of these are about outcomes of negotiations, rather than scientific knowledge gaps. The response to these is not research, but well informed and executed strategy at a number of levels. History tends to suggest that fragmentation in the farming sector, and a tendency to fight for the *status quo* even when this is no longer available, will be the most significant challenges that the industry will have to overcome if it is to best advance its position.

Many of the technical knowledge gaps concern the management of enteric emissions, but beyond this the industry will need to have:

- well-developed farming system approaches to integration of emissions avoidance and carbon sequestration that are far more sophisticated than energy saving and farm forestry. The integration of a range of biofuels with scientifically and institutionally credible sequestration in the soil (including the use of char) will require both technical development, and proof of reliability in use.
- More sophisticated market institutions and products to address the need to integrate many different types of farm-based environmental services with production, including carbon services, biodiversity, water and salinity. Current approaches are fragmented and confusing, *and do not properly recognise the potential social value that can be produced by farming when managed to this end.*
- The farm sector will need to better understand the role of land-clearing in the industrial carbon cycle, and to develop policies that recognise the new reality that land-clearing is no longer only about biodiversity and farming, it is also about the cost of energy and the international positioning of Australia in industrially important negotiations.
- A better understanding of the international trade and competitiveness aspects of carbon markets, including the challenges of trade equality in a world of differing (and possibly non-comparable) carbon market or taxation policies.

If the red meat industries cannot address these complex policy and management issues simultaneously with managing the technical issues, it can be expected that the policy and competitive dynamic which has been discussed will be more hostile than it needs to be.